Eucalypt regeneration and ecological restoration of remnant woodlands in Tasmania, Australia

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Much of the material in Chapter 3 is published as:

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Tanya G. Bailey (70%), Neil J. Davidson (20%), Dugald C. Close (10%).

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Abstract

Lack of eucalypt regeneration is a key factor in the decline of forest and woodland remnants in low rainfall agricultural regions in Australia. This thesis provides a new insight into dry forest and woodland regeneration by demonstrating how important and tightly circumscribed the eucalypt seedling regeneration niche is in these forests in Tasmania. The potential of soil water repellency to be a barrier to eucalypt recruitment and the difficulty of mimicking the regeneration niche to improve natural regeneration processes in degraded forests are highlighted. A section on management implications is included.

Key to effective management of dry forests and woodlands is an understanding of the requirements and conditions that promote seed germination and seedling establishment (recruitment niche) and the persistence of lignotuberous sprouts (juvenile persistence niche). The processes of eucalypt recruitment and persistence were investigated in dry forest and woodland remnants in the Midlands of Tasmania at the scales of the stand and microsite. The work was conducted with a view to applying the results to the discipline of restoration ecology.

Measurements of structural complexity at thirty remnant forest stands that were in a range of stages of decline revealed that healthy remnants contained four times the amount of eucalypt regeneration (seedlings, lignotuberous sprouts and saplings) than stands in a degraded condition and that regeneration was absent in stands of paddock trees (except for one stand in which there had been a fire). An additional lack of trees in smaller diameter size classes in both intermediate and paddock tree sites relative to the amounts shown in healthy sites implies a long-term recruitment scarcity and an unavoidable future bottleneck in the development of mature trees. The quantity of eucalypt regeneration was positively associated with other structural attributes: perennial species and life form richness; cover of vegetation 0.5-6m high; litter; dead trees; large trees; and total length of fallen logs >10cm diameter; and negatively associated with quadratic mean tree diameter at breast height. Across the thirty stands, the majority of eucalypt regeneration was in the form of lignotuberous sprouts and saplings. Only in stands burnt 2-6 years prior to the survey did newly recruited seedlings form part of the regeneration.

Within these burnt sites the seedling recruitment niche differed significantly to the juvenile persistence niche (occupied by lignotuberous sprouts), while the juvenile persistence niche had characteristics similar to the general forest floor. Seedling microsites were characterised by the following: canopy gaps and ashbeds; a predominantly northerly aspect; over 220° shelter in profile; an average distance to a sheltering object of < 30cm; shelter provided by coarse woody debris (80% of seedlings sheltered by logs and branches); soil that was significantly softer and less water repellent than the forest floor; and low cover of grass. All these characteristics of seedling microsites affect moisture availability.

The role of ashbeds, coarse woody debris and soil water repellency in eucalypt recruitment was further investigated. Soil water repellency (hydrophobicity) can be severe in dry eucalypt forests as hydrophobic organic compounds coat the surfaces of soil particles and this has implications for the movement and storage of water in the system. Other studies have shown that soil hydrophobicity is differentially affected by fire depending on the temperatures reached, with repellency increasing as temperature increases until a threshold is reached at which repellency is removed (approximately 260°C). Logs lying on the forest floor provide heavy fuel for intense fire which creates ashbeds. The current study showed that in ashbeds hydrophobicity was removed in surface layers but the hydrophobic layer moved lower (1-3cm) down the soil profile. The wettable surface soil zones enabled the germination of eucalypt seed and subsequent establishment of seedlings. Remaining adjacent and partially burnt coarse woody debris provided a soil moisture store (with threefold the amount of moisture in soil under logs compared to 5m away) and probably protection from microclimatic fluctuations and browsing animals. Surface soil outside of the ashbed areas was severely water repellent suggesting that soil water repellency may be a barrier to eucalypt recruitment in lightly burnt and unburnt soils.

These findings were applied to restoration ecology. Patch scale restoration trials were conducted in six dry forest remnants in the Midlands of Tasmania in an attempt to mimic the eucalypt recruitment and persistence niches through the use of intense spot burns or cultivation with and without the addition of large logs. Survival of seedlings established from introduced seed and natural seed rain was variable within and among sites resulting in no treatment effects. Thus the ‘Burn with the addition of logs’ treatment did not successfully mimic the recruitment niche, probably because

the experiment did not mimic the natural heterogeneity of hydrophobicity following wildfire. However, the survival of planted seedling was greatest in 'Cultivated with no log' treatments, which most closely resembled the juvenile persistence niche. Burning treatments did provide a relatively weed-free seed bed for over a year and surviving seedlings grew significantly better than those in cultivated treatments which quickly became infested with grassy weeds despite granular herbicide application.

Survival of planted seedlings and the amount of observed germination of sown seed were each significantly correlated with the structural complexity score of the planting site with seedling survival significantly higher in sites of healthy condition. This suggests that the more degraded sites had crossed an abiotic threshold (using state and transition model terminology) that was not completely removed by the experimental restoration treatments. The early establishment of planted eucalypt seedlings was also shown to be significantly affected by species; the underlying soil water repellency of the planting plot; the proximity to an adult tree and the type of soil amelioration used in the restoration treatment but not by the presence or absence of logs.

Suggestions for further work include trialling treatments that more closely mimic the recruitment niche by partially burning and retaining logs *in situ* rather than post burn addition; investigating the timing and intensity of weed control in cultivated treatments; testing the effects of soil water repellency amelioration through the use of wetting agents; and investigating spatial aspects of treatment patch placement.

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Chapter 1 Introduction

1.1 Vegetation fragmentation and tree decline

Agriculture and forestry are pervasive, intensive and extensive land uses that have resulted in widespread landscape modification and habitat fragmentation globally (Tilman *et al.* 2001; Foley *et al.* 2005). Habitat loss and fragmentation have had a major adverse impact on biodiversity (the diversity of genes, species, and ecological processes). This in turn has affected the provision of ecosystem services such as healthy soils, water infiltration, nutrient cycling and waste decomposition upon which agriculture and forestry depend (Chapin *et al.* 1997; Tilman *et al.* 2002; Fahrig 2003; Fischer *et al.* 2006). Conservation of biodiversity within production landscapes is therefore of pressing concern (Green *et al.* 2005; Hoekstra *et al.* 2005), not only in reserves set aside for protection (Hobbs and Norton 1996).

Areas once covered by woodland and open dry forests have been particularly affected by vegetation fragmentation due to clearing for agricultural production (Yates and Hobbs 1997a). Indeed the global biome of "Mediterranean forests, woodlands and scrub" (Hoekstra *et al.* 2005), of which the dry temperate sheep/wheat areas of south eastern Australia, Tasmania and south western Australia are a part (Fischer *et al.* 2009), has been rated as being at particularly high risk of biodiversity loss and ecosystem dysfunction because of the disparities between habitat loss and protection in this biome (Hoekstra *et al.* 2005).

The loss of tree cover through clearing and fragmentation of remnant vegetation in agricultural areas has been exacerbated by declines in tree health and lack of regeneration (Reid and Landsberg 2000; Close and Davidson 2004). Suppression of regeneration of dominant tree species has been shown to occur in grazed landscapes

in Australia (Saunders *et al.* 2003; Dorrough and Moxham 2005; Fischer *et al.* 2009; Weinberg *et al.* 2011) and around the world (Pulido *et al.* 2001; Plieninger *et al.* 2003; Palmer *et al.* 2004; Zavaleta *et al.* 2007; Plieninger *et al.* 2011). This phenomenon is of particular concern as remnant trees in agricultural landscapes are "keystone structures" with a disproportionally large number of species and ecological processes relying on them (Manning *et al.* 2006; Fischer *et al.* 2010a). Trees provide shelter to livestock, crops and pastures (Bird *et al.* 1992), prevent soil erosion, acidification and salinisation (Cramer and Hobbs 2002), improve water infiltration (Eldridge and Freudenberger 2005), conserve endemic species and provide habitat for native flora and fauna on a local and landscape scale (Manning *et al.* 2006; Vesk and Mac Nally 2006; Manning *et al.* 2009; Fischer *et al.* 2010a; Fischer *et al.* 2010b).

The low rainfall (<700mm pa) agricultural region of Tasmania (referred to as the Tasmanian Midlands in this thesis) provides a typical example of an area beset by tree decline. The Midlands have been progressively cleared and intensively farmed since European settlement 200 years ago. The remaining native remnant vegetation is highly fragmented into a few large (>50ha) patches (usually on crown land) and numerous medium (50-10ha) and small (<10ha) patches (predominantly on private land) within a matrix of native and improved pasture, cropland, plantations and rural settlements. Most remaining tree cover occurs on unproductive uplands with the more productive lowlands being dominated by small patches and scattered trees (Michaels *et al.* 2010).

The condition of remnant native vegetation is highly variable across the region. Although a small proportion of high-value remnants are set aside as reserves, most

have been subject to degradation by grazing, nutrient enrichment and weed invasion. There is pressing need for conservation of biodiversity in these remnants (Kirkpatrick and Gilfedder 2000; Gilfedder *et al.* 2003; Davidson *et al.* 2007; Michaels *et al.* 2010) as the dry landscapes of the Midlands harbor a disproportionate number of rare, endangered and threatened species (Kirkpatrick and Gilfedder 1995; Gilfedder *et al.* 2003) and communities (Kirkpatrick and Gilfedder 2000; RPDC 2006b).

Eucalypts are the foundation species of the dry forest and woodland communities, playing a vital ecological role structurally and functionally both in remnant vegetation patches and as scattered paddock trees. However, tree decline is a major degrading influence (Neyland 1996; Kirkpatrick *et al.* 2000; Close and Davidson 2004; Davidson *et al.* 2007; Williams *et al.* 2010). More than 50% of the Midlands area has trees in decline (Figure 1-1) with an estimated 30% of the Northern Midlands bioregion suffering from severe to extreme tree decline (Williams *et al.* 2010) .

Tree decline in the Midlands follows the same trajectory as reported in other dry agricultural districts of Australia with dieback of adult trees and lack of seedling recruitment (Yates and Hobbs 1997a; Reid and Landsberg 2000; Gibbons *et al.* 2008b; Fischer *et al.* 2009). Dieback in eucalypts is characterised by "a thinning of the crown that begins at branch ends and progresses towards the trunk. Dead branches typically protrude beyond the remaining foliage. New leaf development subsequently occurs from epicormic buds and may re-occur for years before eventual death" (Close and Davidson 2004) Figure 1-2.

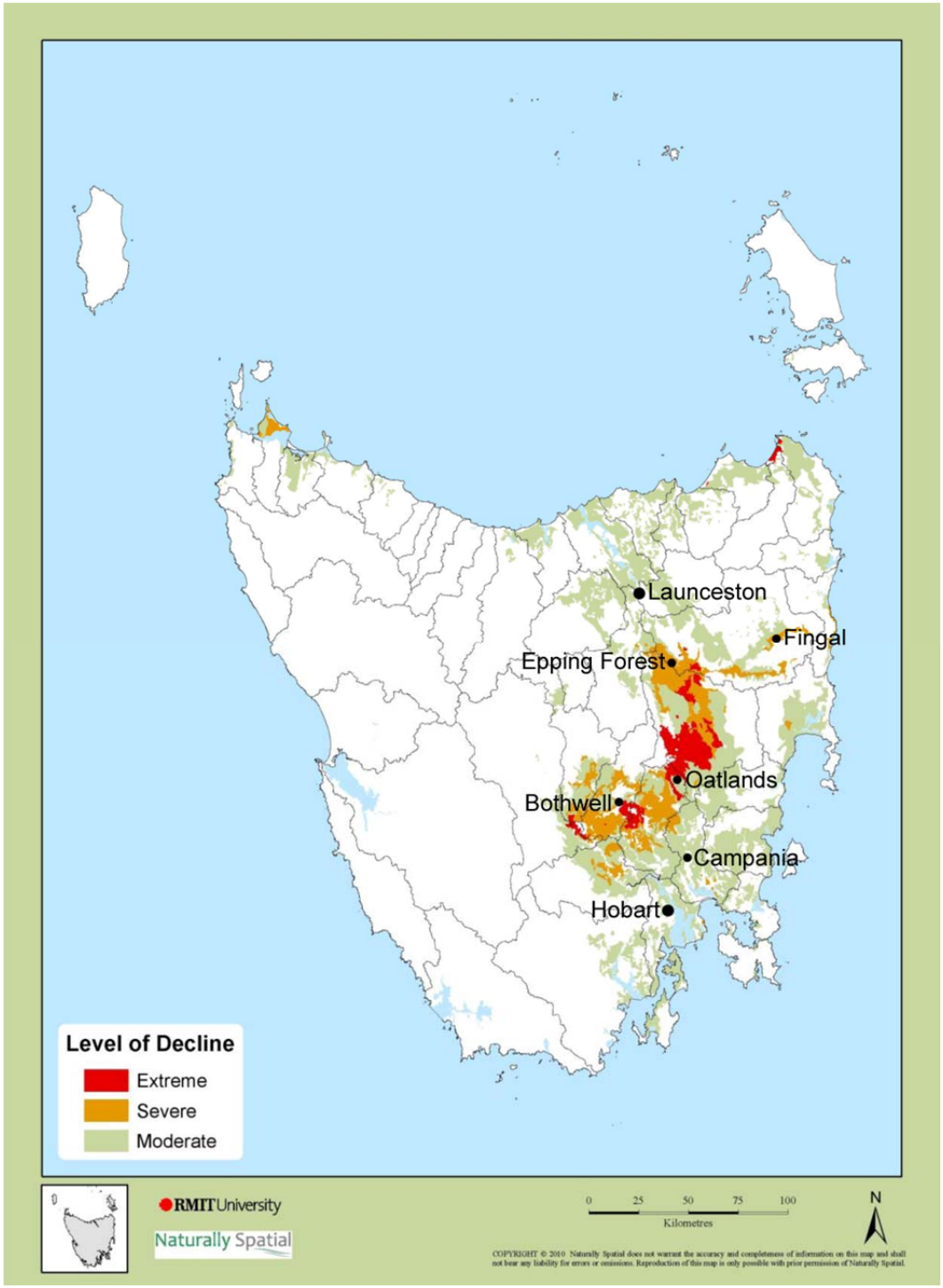

Figure 1-1 The extent and level of tree decline in Tasmania. The Midlands of Tasmania lie between the cities of Launceston and Hobart. The towns in the Midlands near which study sites are located are labelled. Grey lines outline bioregions (Adapted from Williams *et al.* (2010) p 74).

Figure 1-2 Dead and dying paddock trees with no evident eucalypt regeneration in the Midlands of Tasmania.

In a review of tree decline in agricultural landscapes, Close and Davidson (2004) list a range of effects that rural land use have on remnant trees, most of which affect water relations: exposure to increased wind and soil drying; decreased water availability due to competition with improved pasture and decreased infiltration due to soil compaction by cloven-hoofed livestock; increased salinity; loss of ectomycorrhizal fungal diversity; changed nutrient balance due to direct addition of fertiliser and/or nitrogen fixation by clover-based pastures; changed pest/predator and biodiversity balance; and lack of seedling recruitment. These factors in conjunction with below-average rainfall and above-average temperatures over the past three decades are implicated as causes of rural tree decline.

In a study of eucalypt health and agricultural land management within woodland remnants in the Tasmanian Midlands, Davidson *et al.* (2007) found that the health of trees was strongly associated with soil properties such as total nitrogen, pH (both of which were lowest in healthy sites) and organic carbon (highest in healthy sites) as

well as the cover of native shrubs (highest in healthy sites) or exotic pasture species (highest in sites with dead trees Figure 1-2). Grazing history (fencing, grazing frequency and intensity) was the primary management factor separating healthy and poor sites, while patch size, fire frequency and wood collection were secondary but important factors. Close *et al.* (2008), also working in the Midlands, found that eucalypt health was negatively correlated with soil nutrient enrichment, raised foliar nutrient levels and weed invasion, all of which were positively associated with increasing grazing intensity.

Studies in the south eastern Australian temperate grazing region (Dorrough and Moxham 2005; Fischer *et al.* 2009; Weinberg *et al.* 2011) have shown that the probability of a remnant vegetation patch having natural eucalypt regeneration is also influenced primarily by livestock grazing intensity. Other influential factors included size of remnant, public vs. private tenure of the remnant, tree canopy cover (Weinberg *et al.* 2011), exotic annual plant cover, history of cultivation, distance to nearest tree (Dorrough and Moxham 2005), fertilizer use and tree density (Fischer *et al.* 2009). While relatively unmodified remnants (predominantly in public reserves) generally contained eucalypt regeneration (Gibbons *et al.* 2008a; Weinberg *et al.* 2011), eucalypt regeneration was shown to be suppressed near isolated paddock trees as well as in modified remnant patches on privately owned land with a history of heavy to intense grazing (Fischer *et al.* 2009; Weinberg *et al.* 2011).

The viability of remnant stands depends on the balance between the number of individuals removed from the population (through death, clearing or harvesting) and the number entering through recruitment and regeneration, and requires a progression of individuals from smaller to larger size classes (Smith *et al.* 1997; George *et al.*

2005). Modelling by Gibbons *et al.*(2008b) suggested that scattered tree populations will be completely lost from many dry agricultural landscapes within 90–180 years without concerted and sustained effort to reduce tree mortality and increase eucalypt recruitment. Fischer *et al.* (2010b) also predicted a steady expansion in the amount of treeless land over a similar time frame with the most rapid decline in tree cover being in areas in which trees occur at low densities and in introduced pastures, where regeneration is most likely to be absent.

However, in absolute terms, Fischer *et al.* (2010) predicted that a larger number of trees may be lost in areas of higher tree density. These areas may be slowly degrading from woodland patches to scattered trees due to a lack of tree recruitment. Fischer *et al.* (2010) suggest that without remedial management "the proportion of grazing land covered by scattered trees may actually remain relatively constant for some time into the future, whereas the proportion of land with denser tree cover will steadily decline". This has serious consequences for productivity and biodiversity in dry agricultural landscapes as different suites of species are supported by stands of different tree densities (Fischer *et al.* 2010a).

Clearly there is a need to restore the dynamic ecosystem process of regeneration in remnant scattered tree, woodland and forest systems if biodiversity is to be conserved and a treeless landscape is to be avoided in the future. Development of appropriate management and restoration measures demands a better ecological understanding of eucalypt regeneration in these systems and of what restoration techniques are appropriate (Spooner and Allcock 2006).

This falls in the realm of ecological restoration defined here as "intentional human intervention in enhancing ecosystem recovery after disturbance"(Young *et al.* 2005)

with links to the related scientific discipline of restoration ecology. Practical techniques for ecological restoration can and should be informed by ecological principles and concepts (Hobbs and Harris 2001; Young *et al.* 2005; Temperton 2007). With eucalypt regeneration being the focus of this thesis, the ecology of ontogeny (the development of an individual through life) and the concept of the of the regeneration niche (Grubb 1977) are particularly relevant.

1.2 Regeneration niche

The ecological niche is a fundamental concept defined as the set of ecological conditions, both biological (biotic) and physical (abiotic), required for species to develop (Hutchinson 1957). Grubb (1977) and Young *et al.* (2005) suggested analysing ecological niches according to life-history stages of species as niche requirements may change during the life of individuals and ontogenetic constraints are a critical element of population dynamics. In a seminal paper, Grubb (1977) proposed that the regeneration stage of a plant's life cycle was the most crucial in creating niche separation among species as adults often share considerable niche space. Grubb (1977) defined the regeneration niche as "an expression of the requirements for a high chance of success in the replacement of one mature individual by a new mature individual of the next generation".

The regeneration niche of a species has a number of components which relate to the reproductive processes of a plant (Grubb 1977), and Young *et al.* (2005) further refined the definition of the regeneration niche into different ontogenetic niches as follows:

Reproductive niche: the set of environmental parameters that allows adults to produce flowers, engage in successful fertilisation, and rear seeds up to the dispersal stage;

Dispersal niche: the set of environmental parameters that determines where seeds arrive (related to dispersal barriers and limitations);

Recruitment niche: the set of environmental parameters that allows seeds to germinate and become established (safe sites). The recruitment niche could further be divided into separate **germination** and **establishment niches** as the requirements for germination and juvenile survival often differ (Battaglia and Reid 1993; Schupp 1995; Eriksson 2002).

Another life history stage can be added for sprouting species which have storage organs that enable them to resprout after disturbances such as fire or browsing (as do most dry forest and woodland eucalypts). For these resprouters the critical window for survival is the time taken by seedlings to establish viable vegetative propagules, called the "sprouting juvenile period" by Clarke (2000). Once they have developed lignotubers, eucalypts can persist in the landscape as juvenile lignotuberous sprouts for decades (Potts 1986). Sprouting species can also persist as adults though major episodic disturbances such as fire and drought by resprouting from epicormic buds, thus maintaining a long term presence in a community. The set of environmental parameters that allows a plant to maintain its space in a community has been termed the "**persistence niche**" (Bond and Midgley 2001), either juvenile or adult.

A life stage that is part of the regeneration niche acknowledged by Grubb (1977) and Clarke (2000) but not Young *et al.* (2005), and thus not given a niche label, is the stage of advance growth, as juveniles develop and mature to become reproductive

adults (sapling stage in trees). This could be called the "**maturation niche**", defined by the environmental parameters that allow established seedlings to grow to maturity. In communities with juveniles suppressed as sprouts on the forest floor, the maturation niche would determine which and when existing individuals recruit into the canopy (different to the recruitment of new individuals into a population).

Adult niche: The set of environmental parameters that allows established plants to survive and grow. This is called the 'habitat niche' by Grubb (1977) and is likely to be broader than other ontogenetic niches (Young *et al.* 2005) as plants can experience new and more heterogeneous environments as they grow into them (Bazzaz 1991).

The **expressed niche** is the union of all the ontogenetic niches and is the set of environmental parameters in which a population is actually found (Young *et al.* 2005).

Ontogenetic niche shifts occur when the niche requirements change during the life of an individual (Eriksson 2002; Young *et al.* 2005). These shifts reflect changes in resource availability, resource requirements, the ability of an organism to access and use resources, and/or size-dependent biotic interactions such as competition and facilitation (Parish and Bazzaz 1985; Bertrand *et al.* 2011). The concept of ontogenetic niche shifts has been well studied in aquatic and terrestrial animals e.g. (Olson 1996; Dopman *et al.* 2002; Stoffels and Humphries 2003; Takimoto 2003) but less so in plants (but see Parish and Bazzaz 1985; Eriksson 2002; Young *et al.* 2005; Miriti 2006; Quero *et al.* 2008; Bertrand *et al.* 2011) . Plants do not actively seek ideal habitats (as animals may), but they experience distinct differences in tolerances to extrinsic factors at different developmental stages (Parish and Bazzaz 1985). All

stages of a population may reside in a given area and still risk local extinction if conditions required for a single ontogenetic transition are absent. Understanding ontogenetic constraints may be particularly important for managing plant populations in which particular life stages require specific conditions not shared by other life stages (Miriti 2006; Quero *et al.* 2008). Young *et al.* (2005) state that ontogenetic niche shifts "are a useful context for looking at restoration practices" and this will be explored further in this thesis.

Environmental patchiness and "environmental sieves" (*sensu* Harper 1977) are other ecological concepts that are relevant to the concept of ontogenetic niche shifts. Resources are known to be patchy within a landscape at a range of scales (Hobbs and Cramer 2003). Within a forest or woodland stand, patches of ground differ in light, nutrient, and water availability, and in the abundances of seed predators, pathogens, beneficial fungi, herbivores and competitors and thus they differ in suitability for plants (Schupp 1995; Florence 1996; Tommerup and Bougher 2000). Differences in suitability lead to patch-dependent differences in seed survival, germination, establishment and/or subsequent growth and survival. This leads to the conversion of initial landscape patterns of seed source and seed fall into a final landscape pattern of adults (Schupp 1995). Schupp (1995) suggests that this process can be conceived as "the passing of a population through a series of environmental sieves that selectively filter out some individuals while allowing others to pass through to the next stage. More suitable patches allow more individuals to survive and pass through than do less suitable patches. These environmental sieves operate throughout the life of plants from seed to seed-producing adult but in most populations habitat outcomes are imposed primarily during the younger stages where mortality is concentrated (Harper, 1977)". The narrowing of niche space through ontogeny could be viewed as

the equivalent of an environmental sieve (Young *et al.* 2005). This will be explored in the context of soil water repellency in Chapters 4 and 5 of this thesis.

Human activity can temporarily or permanently alter the expressed niche. An example of this is the previously described large scale clearing, fragmentation and degradation of remnant vegetation in dry agricultural landscapes where the recruitment niche has been eliminated in places and eucalypts often embody 'relict populations' (Eriksson 2000) existing as non-recruiting adults. Young et al (2005) suggest that restoration in such situations may require "restoring the lost links in the recruitment chain".

The focus of this thesis is on the eucalypt recruitment and juvenile persistence niches. In order to know which factors are missing in the recruitment niche in degraded stands where regeneration is suppressed, it is necessary to identify the environmental attributes that support the establishment of seedlings in naturally regenerating stands. This will be the focus of the following two chapters with stand scale attributes being assessed in Chapter 2 and small scale (microsite) attributes of the recruitment and persistence niches assessed in Chapter 3.

While acknowledging the importance of the reproductive and dispersal niches in the regeneration of eucalypts in agricultural areas, in depth exploration of these ontogenetic niches are mostly beyond the scope of this thesis. Restoration activities via revegetation often bypass the dispersal niche through the translocation of propagules (seed and or nursery grown seedlings, Young *et al.* 2005). The importance of the reproductive niche in fragmented populations has been explored in a genetics and restoration context elsewhere e.g. (Burrows 2000; Hobbs and Yates 2003; Broadhurst and Young 2007; Krauss *et al.* 2007; Broadhurst *et al.* 2008;

Mimura *et al.* 2009; Ottewell *et al.* 2009; Ottewell *et al.* 2010; Vesk *et al.* 2010; Orscheg *et al.* 2011). There are, however, a number of other concepts that have been utilised by restoration ecologists that are relevant to this thesis and will be described below.

1.3 Restoration ecology

Restoration must address the long lasting reorganisation of ecosystems driven by the landscape modification and vegetation fragmentation legacies of production industries (Suding and Hobbs 2009a). In the field of restoration ecology there has been increasing interest over recent decades in developing better predictive tools and a conceptual framework to guide restoration of degraded land (Hobbs and Norton 1996; Whisenant 1999; Hobbs and Harris 2001; Suding *et al.* 2004; Temperton *et al.* 2004; Young *et al.* 2005; King and Hobbs 2006; Suding and Hobbs 2009b; Suding and Hobbs 2009a; Kardol and Wardle 2010). While early work in the 1980's focused on classical succession theory (Bradshaw 1984; Pickett *et al.* 1987; Luken 1990), models have subsequently been developed that recognise that ecosystem dynamics can be "complex, nonlinear, and often unpredictable" (Wallington *et al.* 2005; Suding and Hobbs 2009a).

Succession models describe restoration as the initiation or acceleration of succession processes along a gradual trajectory of recovery towards a desired state (Luken 1990). More recent models, based on the state and transition approach, suggest that multiple successional pathways are possible and that due to disturbances (such as grazing) different "states" can exist in any particular location (Prober *et al.* 2002; Spooner and Allcock 2006). These state and transition models take into account the following: that the dynamics of a degraded state can be very different from those in a

relatively unmodified state; the trajectory to recovery will most likely be different from that of degradation and; there may be ecological thresholds (biotic and/or abiotic) that serve as barriers to recovery which may only be overcome with management inputs (Hobbs and Norton 1996; Whisenant 1999; Suding *et al.* 2004).

Thresholds are implicated when removal of the degrading influence does not initiate a transition back towards the original state (Hobbs and Norton 1996). Cramer *et al.* (2006) suggest that where a threshold has been passed, the recruitment niche of a native species may no longer exist and this barrier makes the spontaneous regeneration of native species unlikely. Identification of the barrier(s) and some form of intervention or management would be needed to restore suitable conditions for recruitment.

In a paper describing state and transition models in the context of degraded woodlands in south eastern Australia, Spooner and Allcock (2006) state that setting of appropriate restoration goals "requires sound knowledge of the ecosystem, including the desired ''state'' of the ecosystem, and an understanding of limiting and driving processes" and that the identification of barriers to recovery is critical to restoration of degraded woodlands. This may be done by identifying factors (natural events and/or management practices) that shape the characteristics of states and drive transitions between them as well as identifying any biotic and/or abiotic constraints, interactions and internal feedbacks within degraded sites that maintain them in a degraded state (Hobbs and Norton 1996; Suding *et al.* 2004; Spooner and Allcock 2006).

This thesis addresses these issues in the context of restoring regeneration of keystone tree species in remnant vegetation degraded by agricultural practices in the Midlands

of Tasmania. It is acknowledged that re-establishing trees is only the first step of broader ecosystem recovery and that restoration at a landscape scale will also need to include the enlarging of existing remnants, linking existing remnants and creating new stands all of which may involve complementary restoration techniques (Yates and Hobbs 1997a; Close and Davidson 2003; Close *et al.* 2005b; Close *et al.* 2010b).

1.4 Thesis structure and objectives

Chapter 1 provides the general introduction and background to this thesis. Chapters 2-6 are experimental chapters. Chapter 7 synthesises results from previous chapters, presents a conceptual model, identifies research gaps and offers suggestion for the management of remnant woodlands/dry open forests in Tasmania for improved eucalypt regeneration.

The intrinsic motivation of this thesis is to identify and define the critically important attributes of the eucalypt recruitment and persistence niches in the woodland and dry forest communities of the Tasmanian Midlands and to recreate these attributes experimentally in degraded remnants in order to promote eucalypt regeneration. The main objectives of this thesis are:

- to investigate the stand and microsite attributes that support eucalypt recruitment and persistence in dry Tasmanian woodlands
- to explore the affects of soil water repellency on eucalypt recruitment
- to investigate how restoration methods that seek to mimic the attributes of the recruitment and persistence niches in dry Tasmanian woodlands affect eucalypt regeneration in remnant stands of varying condition.

The following is a brief summary of the experimental chapters and the objectives of the particular experiments.

Chapter 2 describes a survey of thirty remnant woodland and dry forest stands in the Midlands of Tasmania ranging in condition from healthy to poor. Thirteen core structural attributes were measured and used to calibrate an index of structural complexity using techniques developed by McElhinny *et al.* (2006) in South-eastern Highlands Bioregion of NSW and ACT and adapted to suit Tasmanian dry open forest/woodland types. The association between the quantity of eucalypt regeneration and other structural attributes, as well as the effect of the history of disturbance of a site were assessed.

Objectives:

- To assess which stand scale structural attributes and types of disturbance history are associated with the absence or presence and amount of eucalypt recruitment.
- To assess in what type of woodland remnants remedial action may be needed to restore eucalypt regeneration.
- To calibrate the McElhinny *et al.* (2006) index of structural complexity for Tasmanian woodlands.

Chapter 3 describes a survey of the attributes of microsites supporting eucalypt seedlings and lignotuberous sprouts in four woodland/dry forest remnants in the Tasmanian Midlands.

Objectives:

- To observe, define and analyse the distinguishing attributes of the microsites that support eucalypt seedlings (recruitment niche) and to establish whether these significantly differ from the general forest floor.
- To observe, define and analyse the distinguishing attributes of the microsites that support eucalypt lignotuberous sprouts (juvenile persistence niche) and to establish whether these significantly differ from the recruitment niche and the general forest floor.

Chapter 4 describes an examination of soil water repellency and moisture around fallen logs at two recently burnt woodland sites at Oatlands and Epping Forest.

Objectives:

- To compare the amount of moisture stored in soil under logs and soil nearby in the general forest floor.
- To measure the soil water repellency at different depths near to and at regular distances from logs post fire.
- To assess the implications of soil water repellency as a barrier to eucalypt seedling establishment.

Chapter 5 describes a patch scale restoration trial which attempted to recreate the eucalypt recruitment niche (as described in Chapter 3) at six sites in the Tasmanian Midlands through the use of burning or cultivation treatments and the addition of large logs.

Objectives:

- To assess the effect of stand condition (measured by structural complexity) on the germination and survival of direct sown and self-sown eucalypt seedlings.
- To assess the effect of establishment treatments on the germination and survival of direct sown and self-sown seedlings.
- To assess barriers to eucalypt recruitment and whether the eucalypt recruitment niche was successfully recreated in the restoration trial.

Chapter 6 describes the second part of the patch scale restoration trial described in Chapter 5. This trial attempted to recreate the eucalypt juvenile persistence niche (as described in Chapter 3) at six sites in the Tasmanian Midlands through the use of burning or cultivation treatments and the addition of large logs.

Objectives:

- To assess the effect of stand condition (measured by structural complexity) on the early survival and growth of planted eucalypt seedlings.
- To assess the effect of establishment treatments on the early survival, growth and health of two local species of planted eucalypt seedlings.
- To assess the affect of soil water repellency on planted seedling survival.
- To assess whether the eucalypt juvenile persistence niche was successfully recreated in the restoration trials.

1.5 Terminology

Various terms are used within ecological literature to describe fragmented remnant vegetation, with some terms being used by different authors to describe very different scales. The terminology used in this thesis will therefore be defined below.

Site: a named study area (e.g. Tom Gibson Reserve), usually a vegetation remnant. There may be multiple remnants and sites within a named study location (e.g. Epping Forest).

Remnant: a relatively continuous area of native vegetation remaining after clearing (called fragments, vegetation patches and bushland remnants elsewhere). Remnants can vary greatly in size, shape and condition and can vary from one to many stands.

Stand: an area of trees with similar structural and floristic characteristics. A stand is the scale at which structural complexity measurements reported in Chapter 2 are made. Stands have multiple and heterogeneous patches within them.

Patch: an area of ground within a stand that may or may not contain trees. A patch is the scale at which restoration treatments are undertaken in Chapters 5 and 6. Patches contain multiple microsites.

Microsite: small scale point on the ground (e.g. in which seeds can lodge and seedlings establish). Seedling centred microsites are the focus of Chapter 3.

Woodlands: The dry sclerophyll communities of the Midlands are distinguished by the dominant eucalypt species in the canopy in combination with the type of environment in which they occur. In many cases in Tasmania, woodland (10-30% projected foliage cover) may grade into open forest (30-70% projected foliage cover) which are floristically similar with only tree densities dividing them (Harris and Kitchener 2005). The tree densities in remnant vegetation often change irregularly (Harris and Kitchener 2005) and woodland/ open forest boundaries can be difficult to determine (Gilfedder *et al.* 2003). The term woodland will be used predominantly in this thesis for reasons of simplicity. While the majority of remnants studied would be classified as woodlands some may have had areas of open forest and results apply to both. General terms used such as "forest floor" also apply to woodlands.

Chapter 2 Stand structural complexity and eucalypt regeneration

2.1 Introduction

Natural eucalypt regeneration has been shown to occur in relatively unmodified remnant woodland vegetation but to be highly restricted in modified remnants and grazed pastures in many dry agricultural districts of Australia. This includes the south eastern temperate grazing region (Nadolny 1995; Dorrough and Moxham 2005; Spooner and Briggs 2008; Fischer *et al.* 2009; Gibbons *et al.* 2010; Weinberg *et al.* 2011); the West Australian wheat belt (Yates *et al.* 1994a; Norton *et al.* 1995; Saunders *et al.* 2003); the Mt Lofty Ranges bioregion of South Australia (Ottewell *et al.* 2010); the ACT (Landsberg *et al.* 1990) and the Northern Tablelands of NSW (Curtis 1990; Landsberg *et al.* 1990). While a number of studies have assessed the condition of woodland and dry forest remnants in the dry agricultural district of Tasmania, the focus of these studies was not on eucalypt regeneration but rather on tree health (Davidson *et al.* 2007; Close *et al.* 2008); age, shape, disturbance and proximity of remnants (Gilfedder and Kirkpatrick 1998); rare and threatened species (Kirkpatrick and Gilfedder 1995); bryophyte diversity (Pharo *et al.* 2005); and bird species composition and richness (MacDonald and Kirkpatrick 2003). This current study focuses on the relationship between remnant vegetation condition and eucalypt regeneration.

There are many measures of 'health' or 'condition' of remnant vegetation that attempt to integrate the range of attributes (including regeneration) that contribute to biodiversity. These are generally qualitative and have their critics because of this. For example Gibbons and Freudenberger (2006) consider 'condition' to be a "value-

laden concept that requires data to be interpreted through a 'value prism' along a continuum from 'good to 'bad'." Keith and Gorrod (2006) suggested that vegetation condition has three main facets with values based around aesthetics, production and biodiversity and that the context, meaning and scope of 'condition' needs to be explicitly articulated in each application. For the purposes of this thesis vegetation condition is being defined within the context of biodiversity values, that is, the "capacity of native vegetation to sustain local populations of native plants and animals" (Keith and Gorrod 2006), with a particular focus on eucalypts.

McElhinny *et al.* (2006) suggest that 'condition' (used in the context of biodiversity) should be measured quantitatively at a stand, or "site" scale. This is also the scale at which restoration efforts are most likely to be made (Yates and Hobbs 1997a). Assessments of condition made at individual sites need to be compared to each other and to the range of conditions across a study area, as condition is a relative rather than absolute concept (Gibbons and Freudenberger 2006). Therefore, an index which combines data from multiple attributes into a single score is a useful tool. Indices allow sites to be ranked according to their assessed condition and for their potential contribution to biodiversity (Parkes *et al.* 2003; McElhinny *et al.* 2005). They may also be useful as a monitoring tool over time, especially at sites that undergo restoration treatments or a change in management regime (Hobbs and Norton 1996).

The method used by government agencies to assess vegetation condition in Tasmania (Michaels 2006) has been adapted from the 'Habitat Hectares' method developed by Parkes *et al.* (2003) and involves assessing site-based and landscape components against a benchmark defined for a particular TASVEG vegetation community (as described and mapped in the TASVEG statewide mapping, Harris and Kitchener

2005). The vegetation condition and landscape context scores are added to produce a single condition score which gives an indication of the degree to which the site differs from the 'natural' or benchmark state (Parkes *et al.* 2003; Michaels 2006).

The 'Habitat Hectares' assessment method is subjective, does not provide continuous quantitative data on the amount of regeneration present at a site and therefore makes it difficult to relate regeneration to other vegetation attributes and management regimes. A benchmark approach is also problematic as it does not take into account that a particular vegetation type may have a range of 'natural' stable states (McCarthy *et al.* 2004).

During his PhD studies, Chris McElhinny from Australian National University developed a structural complexity index that was based on the quantitative measurement of a range of attributes in dry eucalypt woodland and forests. Structural complexity is a measure that integrates the range of micro-environments (or microhabitats) available to organisms. The greater the range of structural components in an ecosystem, the greater the variety of resources (and microhabitats) and thus a greater number of plants and animals can utilize these resources (Lindenmayer *et al.* 2000; McElhinny *et al.* 2005; Fischer *et al.* 2006).

The structural complexity index was not based on expert opinion or any idea of what a "natural" state is. It is mathematically and conceptually simple and is based on the range of condition in a region of interest (McElhinny 2005). A comprehensive review of the links between structural attributes, habitat provision and biodiversity (McElhinny *et al.* 2005) and the methodology of constructing the index (McElhinny *et al.* 2006) have been reported.

McElhinny (2005) initially measured a comprehensive suite of over 70 structural attributes at 48 sites across the South-eastern Highlands Bioregion in the ACT and NSW. The set was reduced to 13 core attributes through redundancy analysis of correlations between measured attributes. The aim was that each core attribute had low kurtosis (peakedness) in its distribution among sites, distinguished effectively between site types (woodland and dry sclerophyll forest), functioned as a surrogate for other co-related attributes and was efficient to measure in the field. Each core attribute contributed a potential 10 points, the scores for each attribute were added, and then the total expressed as a percentage of the mathematical maximum (130). A sensitivity analysis revealed that there was no need for weighting of attributes and thus the final index is a simple, additive, non weighted score that rates forest or woodland at each site relative to the range of observed attribute levels across all sites. Being able to assess each individual attribute relative to the range of reference values also provides a useful method of determining whether remedial action needs to be taken to improve that attribute at a particular site. For use in regions outside that in which it was developed, the index would need to be calibrated to a new set of reference sites in the region of interest. This would maintain the ability of the index to distinguish between sites (pers.comm C.McElhinny).

The McElhinny index of structural complexity was calibrated to Tasmanian conditions and used in this study to assess the condition of dry forest and woodland remnants, with particular reference to eucalypt regeneration and the stand attributes associated with it. The research questions examined were:

1. What stand scale factors are associated with eucalypt regeneration in remnant vegetation?

2. What levels of condition and disturbance of remnants support or inhibit eucalypt regeneration?
3. How can the McElhinny *et al.* (2006) structural complexity index be calibrated to the range of Tasmanian dry forest and woodland conditions and how effective were decisions made in calibrating the index for such purpose?
4. Is the observed quantity of eucalypt regeneration sufficient or is remedial management needed to improve eucalypt regeneration in Tasmanian dry forests and woodlands?

2.2 Methods

2.2.1 Sites

Research sites were located in lowland remnant woodland and forest stands across the lower Derwent Valley, Coal Valley and Midlands of Tasmania (collectively called the Midlands from here on, see figure 1-1). The study area was low altitude (below 470m), low rainfall (< 700 mm annual average), extending from latitude 41.511° to 42.622°S and longitude 146.958° to 147.984°E. In summer, maximum temperatures average 24°C, whilst in winter most days have frosts, sometimes severe. Thirty sites were studied (Table 2-1) comprising remnant woodland and forest stands which had been exposed to a variety of levels of disturbance and recent fire history. The sites were allocated *a priori* and equally by number to one of three ranked categories of vegetation condition namely healthy, intermediate and poor according to the following features (adapted from Keighery (1994):

- healthy – all expected plant layers present and healthy, very light to moderate levels of grazing and wood collection, any disturbance confined to small areas, few or localised exotic species

- intermediate – shrub and ground layer present but reduced of simplified, moderate to heavy levels of grazing and wood collection, apparent change in soil structure, exotic species cover up to 50% of area
- poor – plant community severely altered, evidence of heavy to extreme grazing, predominantly paddock trees in pasture. Most poor sites were less than 500m away from other more substantial remnant patches and were usually separated from them by fencing.

Sites were rated for levels of disturbance caused by grazing as light (< 1 dry sheep equivalent (DSE)), moderate (1-2 DSE), heavy (2-4 DSE) and extreme (>4 DSE). Their recent fire history (within last 10 years) was obtained from land managers of each site (Table 2.1). Four sites were publicly owned reserves, five private reserves and the rest on privately owned unreserved land. Six sites were near Bothwell and one in Elderslie in the Derwent Valley (*Eucalyptus tenuiramis* woodland*)*, eight sites were near Oatlands in the southern Midlands *(E. pauciflora /E. viminalis* woodland*)* and twelve sites were in the northern Midlands, eight near Epping Forest (*E. amygdalina/ E. viminalis* woodland) and four near Fingal (*E. amygdalina/ E. viminalis* woodland). Three sites were in the Coal valley near Campania (*E. amygdalina/ E. viminalis* woodland), see Figure 1-1. The sites at Oatlands and Bothwell had all been previously established for research reported by Davidson *et al* (2007), two sites at each of Epping Forest, Fingal and Oatlands were measured prior to restoration treatments being undertaken as part of a trial reported in Chapter 5 of this thesis and four others were measured prior to a seedling microsite study reported in Chapter 3. The other sites were chosen to ensure a range of *a priori* remnant condition was sampled across a broad geographical range in the Midlands.

Table 2-1 Thirty sites with site name abbreviation, location, *a priori* health class, disturbance level, recent fire history, dominant eucalypt species and TASVEG community (Harris and Kitchener 2005) that have been measured to provide base data for a structural complexity index for Tasmanian dry forests/woodlands

Site(abbreviation)	Location	Class	Disturbance	Fire history	Canopy dominant	TASVEG
Ellis Private Reserve (BOTH)	Bothwell	Healthy	Light	burnt 2005	*E. tenuiramis*	DTO
Humbie Hill Private Reserve (HH)	Bothwell	Healthy	Moderate	burnt 2002	*E. tenuiramis*	DPD
Elderslie Nature Reserve (ELD)	Elderslie	Healthy	Light	burnt 2006	*E.tenuriamis*	DTO
Gravelly Ridge Conservation Area (GR)	Campania	Healthy	Light	burnt 2002	*E. tenuiramis*	DTO
Lowdina Regrowth (LR)	Campania	Healthy	Moderate	unburnt	*E.amygdalina*	DAS
Western Tom Gibson Reserve (WTG)	Epping Forest	Healthy	Light	unburnt	*E.amygdalina*	DAZ
Tom Gibson Reserve Burnt (TG)	Epping Forest	Healthy	Light	burnt 2003	*E.amygdalina*	DAZ
Barton Farm Private Reserve (B)	Epping Forest	Healthy	Moderate	burnt 2004	*E.amygdalina*	DAZ
Evercreech coupe (EC)	Fingal	Healthy	Light	unburnt	*E.amygdalina*	DAM
Radio Mast Hill, Plot 1Weedings (W1)	Oatlands	Healthy	Light	unburnt	*E.pauciflora*	DPO
Meaburn Peratta Tip Reserve , (MPT)	Oatlands	Intermediate	Moderate	unburnt	*E.pauciflora*	DPO
Humbie Intermediate (HI)	Bothwell	Intermediate	Moderate	unburnt	*E. tenuiramis*	DPD
Campbell Intermediate (CI)	Bothwell	Intermediate	Heavy	burnt 2005	*E. tenuiramis*	DTO
Lowdina Simple (LS)	Campania	Intermediate	Heavy	unburnt	*E.amygdalina*	DAS
Downey Valleyfield remnant (DD)	Epping Forest	Intermediate	Heavy	partially burnt	*E.amygdalina*	DAZ
Osbourne remnant (O)	Epping Forest	Intermediate	Moderate	unburnt	*E.amygdalina*	DAZ
Evercreech plantation remnant (ED)	Fingal	Intermediate	Heavy	unburnt	*E.amygdalina*	DAM
Meaburn Liliesleaf remnant (M)	Oatlands	Intermediate	Heavy	unburnt	*E.pauciflora*	DPO
Weedings Plot 2 Bald Hill (BH)	Oatlands	Intermediate	Heavy	unburnt	*E.pauciflora*	DPO
Weedings Plot 3 (W3)	Oatlands	Intermediate	Heavy	unburnt	*E.pauciflora*	DPO
Campbell Paddock Tree (CPT)	Bothwell	Poor	Extreme	burnt 2005	*E. tenuiramis*	DTO
Humbie Paddock Tree (HPT)	Bothwell	Poor	Extreme	unburnt	*E. tenuiramis*	DPD
Downey Paddock Tree 1 (DPT1)	Epping Forest	Poor	Extreme	unburnt	*E.amygdalina*	DAZ
Downey Paddock Tree 2 (DPT2)	Epping Forest	Poor	Extreme	unburnt	*E.amygdalina*	DAZ
Downey Paddock Tree 3 (DPT3)	Epping Forest	Poor	Extreme	unburnt	*E.amygdalina*	DAZ
Evercreech Paddock Tree 1 (EPT1)	Fingal	Poor	Extreme	unburnt	*E.amygdalina*	DAM
Evercreech Paddock Tree 2 (EPT2)	Fingal	Poor	Extreme	unburnt	*E.amygdalina*	DAM
Oatlands Paddock Tree 1 (PT1O)	Oatlands	Poor	Extreme	unburnt	*E.pauciflora*	DPO
Oatlands Paddock Tree 2 (PT2O)	Oatlands	Poor	Extreme	unburnt	*E.pauciflora*	DPO
Oatlands Paddock Tree 3 (PT3O)	Oatlands	Poor	Extreme	unburnt	*E.pauciflora*	DPO

The remnants measured in this study fell into six TASVEG vegetation communities (Table 2-1) all within the same vegetation type called "Dry Eucalypt Forest & Woodland Vegetation" (Harris and Kitchener 2005). The TASVEG benchmarks for the six vegetation communities were very similar in their structural components with the main differences being in their expected species complements. All the communities measured were therefore combined in the same structural complexity index. The effectiveness of this decision was tested through statistical analyses (see section 2.2.5).

Stands of paddock trees were included in this study, and thus incorporated into the Tasmanian index, because they are important biological legacies in the landscape and they account for a substantial amount of remnant woody vegetation cover in lowland areas that have been significantly cleared for agriculture (Gibbons and Boak 2002; Gibbons *et al.* 2008b; Manning and Fischer 2010). Scattered paddock trees provide ecosystem functions such as habitat at the local and landscape scale and may be foci for restoration efforts in the future (Reid and Landsberg 2000; Manning *et al.* 2006).

2.2.2 Sampling design

In each stand three quadrats were established along a transect that ran diagonally across the prevailing slope, passing through the centre of the stand, following the method of McElhinny (2005) (Figure 2-1). The first quadrat began at a random distance between 1 and 100m from the edge (y in Figure 2-1) and the second and third quadrats at equal distances from each other and the end of the transect (x in Figure 2-1). The latitude and longitude of the start and finish of each transect and quadrat were recorded using a handheld GPS unit. Each of the three quadrats was

50m long by 20m wide with the length running along the transect in order to sample the main environmental gradient (McElhinny *et al.* 2006).

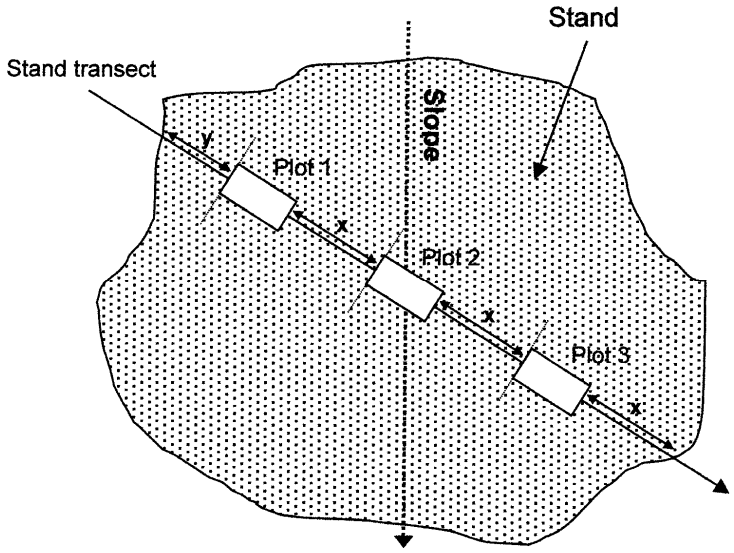

Figure 2-1 Stand transect position in relation to prevailing slope direction. (Source: McElhinny (2005):85)

Each 50m x 20m quadrat contained a 20m x 20m sub-quadrat which was further divided into 10m x 10m sub-quadrats for vegetation cover estimation (Figure 2-2).

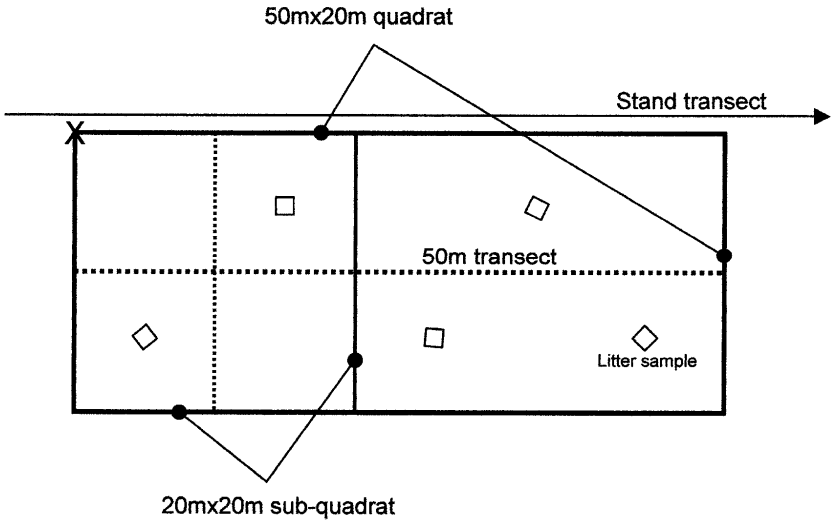

Figure 2-2 Arrangement of 50m x 20m quadrat along stand transect with 20m x 20m sub-quadrat further divided into four 10m x 10m sub-quadrats and positions of typical litter sampling points (Source: McElhinny (2005):86).

Stands of scattered paddock trees were measured so that each quadrat contained at least two trees with at least one of these being alive. The transect through the stand was therefore not always in a straight line with quadrats going off at an angle to include at least two trees. At most paddock tree sites the number of trees per hectare could be counted and thus the sample estimates given by the means of the three quadrats could be compared directly with the total count per hectare and thereby checked for accuracy and representativeness.

2.2.3 Attributes sampled

Ideally the full comprehensive suite of over seventy structural attributes identified by McElhinny (2005, Appendix 2 p206) would have been measured at the Tasmanian sites and then a similar process of reduction to a core set been undertaken. However, time and budget restrictions prevented this and developing new methodology was not the focus of this thesis. The woodland vegetation type was considered structurally and floristically similar enough in the South-eastern Highlands Bioregion and Tasmanian Midlands to assume that a similar, if not exactly the same, set of attributes would have been arrived at if the full process had been undertaken (pers.comm. C.McElhinny).

Attributes sampled were:

In 20m × 20m sub quadrat

- Perennial species richness
- Life form richness divided into the following 10 categories: tussock grass, non-tussock grass, low shrub 0-0.5m, tall shrub >0.5m, graminoids, ferns, vines, overstorey regeneration <2m, overstorey regeneration >2m, trees.

In four 10m × 10m sub quadrats

- Vegetation cover < 0.5m in height: % cover estimated by taking the mean of four 10m × 10m sub quadrats
- Vegetation cover 0.5-6m in height: % cover estimated by taking the mean of four 10m × 10m sub quadrats

In 50m × 20m quadrat

All live trees >5cm diameter were measured for diameter breast height over bark (dbh) in centimetres, 1.3m above ground using a diameter tape with this data being used to derive the following:

- stand basal area (m^2 ha^{-1}) using the calculation

  $BA = \sum \pi (dbh^2 / 40000)$
- quadratic mean dbh of live stems (cm) using the calculation

  $$dbh_Q = \sqrt{\frac{\sum dbh^2}{N}}$$ where N is the number of stems
- number of live stems >40cm dbh ha^{-1}.

Counts were made of the following and data were converted into per hectare basis:

- number of hollow bearing trees
- number of dead trees
- number of regenerating eucalypt stems <5cm dbh (seedlings, lignotuberous sprouts and saplings).

Measurements of the length of all coarse woody debris with a diameter of 10cm or more were made to the nearest 0.5 m and used to calculate:

- total log (≥ 10cm diameter) length (m ha^{-1})
- total large log (> 30cm diameter) length (m ha^{-1}) - a subset of the total log length.

Five litter samples were collected from all dead organic matter <10cm diameter, within a 50cm x 50cm quadrat thrown sequentially left and right at systematic

distances along the central transect (see Figure 2-2). The samples were dried in an oven at 65^{o}C for at least 48 hours, weighed and total sample weights used to derive an estimate of:

- dry weight of litter (t ha^{-1}).

2.2.4 Calibration of structural complexity index

The mean of the three quadrats measured at each site was calculated for each of the 13 core attributes and these values were used to calibrate the structural complexity index to Tasmanian dry woodland conditions, according to the following steps, as per McElhinny *et al.* (2006).

1. The means from the 30 sites were tested for a) normality of distribution for each attribute, using the Shapiro-Wilks (null hypothesis of a normal distribution) and b) kurtosis using the statistics package R (R Development Core Team 2008). Transformations to improve normality were made where necessary. Each transformed attribute was then checked again for kurtosis. A requirement for McElhinny (2005) to include an attribute in his index was low or negative kurtosis, with a kurtosis >2 regarded as high, limiting the potential of an attribute to distinguish between sites.
2. The raw data was rescaled as a 0-10 score by fitting linear regressions to the quartile mid points (12.5, 37.5, 62.5, and 87.5 %) of attribute distributions with scores of 2.5, 5, 7.5 and 10 given to the related quartile midpoint. The maximum score of 10 was set at 87.5% to prevent extreme values distorting the scaling of scores. The equations were constrained so that the rescaled score was always between 0 and 10 and a zero score in the raw data always resulted in a zero score in the rescaled data. This rescaling process resulted in

each site being scored relative to the range of observed attribute levels across all sites.

3. Correlations, using Pearson's product-moment correlation test in R, were run between original and rescaled data to test whether there was much information lost in the rescaling process.
4. Using the template of the Excel file that is McElhinny's index, half of the formulae were removed (as the index was organised for two different vegetation types) and were replaced by the new equations based on the rescaled Tasmanian data. Each of the rescaled attributes was not weighted and accounts for 10 points, adding to a total of 130. The index is expressed as a percentage.
5. The distribution of the index scores was checked for normality of distribution. The capacity of the index to distinguish between sites and between *a priori* remnant condition classes was tested.

2.2.5 Statistical analysis of attribute data

A three dimensional ordination of the unscaled data from the thirty sites was produced using semi-strong-hybrid-multidimensional scaling (SSHMDS) based on Bray-Curtis dissimilarities in PATN (Belbin 2004) . This analysis was performed with a random starting configuration on all 13 measured variables after they were standardized within the range of 0-1 by [(Value-Minimum Value)/ Range of Values]. Principal component correlation (PCC) vectors were fitted to the ordination using multiple linear regression in PATN to determine the relationship between measured attributes and the patterns in the ordination. The robustness of the PCC results and significance of these vectors ($p<0.01$) were tested using a Monte-Carlo approach.

Stress was calculated as a measure of how well the objects had been fitted into a reduced dimensional space by the ordination (Belbin 2004).

Analyses of variance (ANOVA) using R were run to check whether any of the attributes differed significantly among community types and locations. This represents a test of whether the decision to assess all the dry eucalypt forest and woodland communities across the Midlands according to a single index of structural complexity was reasonable. The spread of location and community types in the SSHMDS ordination were also checked through colour coding of sites.

Correlation tests were run between the overstorey regeneration attribute [ln (number of regenerating stems ha^{-1} +1)] and the other 12 attributes in order to determine whether regeneration is associated with any other measured structural attributes. ANOVA s were also undertaken to determine whether the amount of regeneration was different between the *a priori* remnant condition groups (healthy, intermediate, poor); between burnt and unburnt remnants; and between remnants with different non-fire disturbance histories.

The ratio of regenerating to adult stems was determined for each site by dividing the number of regenerating stems (<5cm diameter) per hectare by the number of stems ≥5cm diameter per hectare. The average, range and number of sites with a ratio greater than 1 was calculated for each remnant condition group in order to determine where recruitment is occurring at less than individual replacement rates.

ANOVA was undertaken on all structural attributes to check for differences between previously assigned (*a priori*) remnant condition groups.

2.3 Results

2.3.1 Attributes

The mean values for seven out of the thirteen attributes in this Tasmanian study fell between the two studies conducted by McElhinny (2005) in South-eastern Highlands Bioregion (SHB) of NSW/ACT. The key differences between the studies related to higher means in vegetation cover, quadratic mean dbh, number of large trees and log lengths in Tasmania (Table 2-2).

Table 2-2 Ranges and means of 13 structural complexity attributes measured at 30 sites in the Midlands of Tasmania (TAS) compared with the means of the woodland and sclerophyll vegetation types measured by McElhinny *et al.* (2006) in the South- eastern Highlands Bioregion (SHB) of mainland Australia.

Attribute	TAS min	TAS max	TAS mean	SHB woodland mean	SHB dry sclerophyll mean
Vegetation cover < 0.5m (%)	2.9	94.6	68.9	54	29
Vegetation cover 0.5m-6m (%)	0	55.8	8.5	2.3	4.4
Perennial species richness per 400 m^2	4	25.7	12.9	12.5	20.1
Life form richness per 400m^2	3	9.7	6.5	6.3	8.3
Basal area (m^2 ha^{-1})	9.9	41.1	23.5	16.8	27.0
Quadratic mean dbh (cm)	19	141.2	69.2	37.9	24.8
Number of regenerating stems ha^{-1}	0	963.3	165.7	149	177
Number of hollow bearing trees ha^{-1}	3.3	36.7	18.1	5.2	27.4
Number of live stems ha^{-1} > 40cm dbh	16.7	123.3	40.3	28.7	35.7
Number of dead trees ha^{-1}	0	206.7	38.7	7.3	50.5
Total log length (m ha^{-1})	141.7	2043.3	912.5	182	519
Total large log length (m ha^{-1})	0.2	391.7	172.3	34.5	35.7
Litter dry weight (t ha^{-1})	1.7	21.2	10.4	6.8	14.3

Higher levels of vegetation cover 0-0.5m height and bigger trees in the Tasmanian context may reflect the inclusion of paddock tree sites which tended to have very large trees surrounded by high levels of pasture cover (both native and introduced).

A quite remarkable difference is the much greater length of logs (coarse woody debris) found in Tasmanian dry woodlands compared to those mainland sites measured by McElhinny *et al.* (2006). On average, a fivefold greater log length was found in Tasmanian than in SHB woodlands and 1.75 times the amount in SHB dry sclerophyll forests. A nearly fivefold greater amount of large logs was measured in Tasmania compared to both site types in the SHB (Table 2-2).

2.3.2 Calibration of the structural complexity index for Tasmania

1. **Attribute distributions and transformations**

The 13 core attributes used to calibrate the index were:

- Vegetation cover < 0.5m (%)
- ln(% Vegetation cover 0.5-6m+1)
- ln(Perennial species richness per 400m^2)
- Life form richness per 400m^2
- Stand basal area (m^2 ha^{-1})
- ln(Quadratic mean dbh(cm))
- ln(Number of regenerating stems ha^{-1} +1)
- Number of hollow bearing trees ha^{-1}
- ln(Number of live stems ha^{-1} >40cm dbh)
- ln(Number of dead trees ha^{-1} +1)
- Total log length ha^{-1} (over 10cm diameter)
- Total large log length ha^{-1} (over 30cm diameter)
- Litter dry weight t ha^{-1}

Normality tests revealed the need for transformation of six variables. Taking the natural logarithm of these attributes (plus 1) improved normality of the distribution and reduced kurtosis compared to untransformed data. However vegetation cover 0.5-6m high and number of regenerating stems per hectare had high numbers of zeros (13 and 10 out of 30 respectively) which skewed their distributions and

remained nowhere near normal even after transformation (Shapiro-Wilks p values of <0.001). Transformation did reduce their kurtosis to below 2 (1.5 and 1.4 from 6.7 and 4.7 respectively).

Quadratic means of tree diameter (dbh) had an almost bimodal distribution with the paddock tree sites having only a few large trees while the healthier sites tended to have more small trees. Log- transforming this attribute did improve normality with the p value in the Shapiro-Wilks test increasing from 0.003 to 0.024 and kurtosis was below 2.

The normality of the distribution for vegetation cover < 0.5m high attribute was compromised by an outlier from Elderslie Nature Reserve which had very low cover (2.9%) due to an intense fire two years previously. Removing the outlier resulted in a normal distribution with a mean of 71.1% and kurtosis of 2.7. However as the data point represented a valid ecological measurement it was retained, reducing the Shapiro-Wilks p value to 0.0012 from 0.540. Transformation failed to improve normality of this attribute and kurtosis was high (7.1).

2. Scaling of attributes

Regression equations were fitted to the 12.5, 37.7, 62.5 and 87.5 percentiles of each attribute distribution, which had been awarded a score of 2.5, 5, 7.5 and 10 respectively. The quartile midpoints of the 13 attribute distributions are shown in Table 2-3 and the set of equations used for scoring the 13 attributes are shown in Table 2-4. The ln (% Vegetation cover 0.5-6m+1) attribute regression was marginally non significant (p=0.052) probably due to both of the first two quartile midpoints being zero (Tables 2-3 and 2-4), which reflects the large numbers of zeros in the raw data. However this attribute and equation were retained in the index. The rescaling process is illustrated in Figure 2-3 with two attribute examples.

Table 2-3 Quartile midpoints were used to rescale attribute data as a score from 0 -10. The quartile midpoints corresponded to the 12.5, 37.5, 62.5 and 87.5 percentiles of the attribute distribution and were awarded a score of 2.5, 5, 7.5 and 10 respectively following McElhinny (2005).

		* Quartile midpoints			
Attribute	Percentile	12.5	37.5	62.5	87.5
	(Score)	(2.5)	(5)	(7.5)	(10)
Vegetation cover < 0.5m (%)		52.9	67.7	76.4	84.1
ln(% Vegetation cover 0.5m-6m +1)		0	0	6.3	21.1
ln(Perennial species richness per 400 m^2)		5.1	9.2	13.5	23.1
Life form richness per 400m^2		3.9	6.3	7.3	8.3
Basal area (m^2 ha^{-1})		14.5	20.4	24.9	31.9
ln(Quadratic mean dbh (cm))		30.5	41.5	91.0	122.8
ln(Number of regenerating stems ha^{-1} +1)		0	3.3	86.2	479.2
Number of hollow bearing trees ha^{-1}		12.1	13.3	20	26.7
ln(Number of live stems ha^{-1} >40cm dbh)		20.8	30	37.1	61.1
ln(Number of dead trees ha^{-1} +1)		5.4	13.3	27.1	75.8
Total log length (m ha^{-1})		417.0	693.1	974.4	1522.5
Total large log length (m ha^{-1})		47.1	138.3	192.1	297.9
Litter dry weight (t ha^{-1})		5.4	9.3	10.8	15.0

Table 2-4 Regression equations for scoring attributes on a scale of 0-10 as a function of the raw attribute data. Regression equations were fitted to the 12.5, 37.7, 62.5 and 87.5 percentiles of each attribute distribution, which had been awarded a score of 2.5, 5, 7.5 and 10 respectively. The r^2 and p values of each regression equation are shown. Each equation was constrained so that the attribute score was always between 0 and 10, following McElhinny et al (2005).

Attribute	Regression equation	r^2 p value
Vegetation cover < 0.5m (%)	score= -10.087 + 0.233*(% Vegetation cover < 0.5 m)	0.97 0.015
ln(% Vegetation cover 0.5m-6m +1)	score= 3.717 + 1.991* ln(% Vegetation cover 0.5-6m +1)	0.90 0.052
ln(Perennial species richness per 400 m^2)	score= -5.815 + 5.034*ln(Perennial species richness)	0.99 0. 003
Life form richness per 400m^2	score=-4.439+ 1.660*(Life form richness)	0.95 0.026
Basal area (m^2 ha^{-1})	score= -3.732 + 0.435*(Stand basal area)	0.99 0. 003
ln(Quadratic mean dbh (cm))	score= -13.721 + 4.852* ln(Quadratic mean dbh)	0.96 0.018
ln(Number of regenerating stems ha^{-1} +1)	score= 2.799 + 1.141 *ln(No. of regenerating stems ha^{-1} +1)	0.98 0.009
Number of hollow bearing trees ha^{-1}	score= -2.067 + 0.462*(No. of hollow bearing trees ha^{-1})	0.93 0.033
ln(Number of live stems ha^{-1} >40cm dbh)	score= -19.546+ 7.228* ln(No. of stems ha^{-1} >40 cm dbh +1)	0.97 0.012
ln(Number of dead trees ha^{-1} +1)	score= -2.9336 + 3.0212*ln(No. of dead trees ha^{-1}+1)	0.98 0.003
Total log length (m ha^{-1})	score= 0.189+ 0.007*(Total log length ha^{-1})	0.97 0.017
Total large log length (m ha^{-1})	score= 1.087 + 0.0306* (Total large log length ha^{-1})	0.97 0.017
Litter dry weight (t ha^{-1})	score= -1.84246 + 0.79802*(Litter dry weight)	0.97 0.014

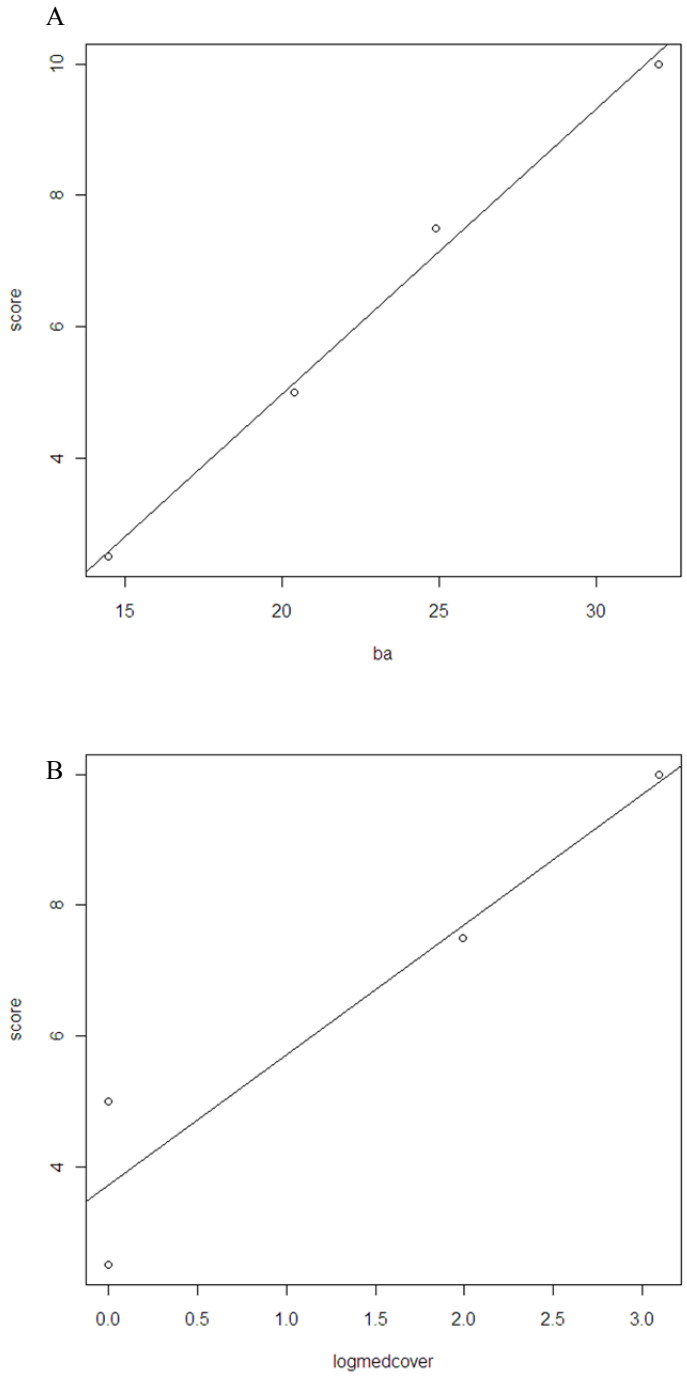

Figure 2-3 Two examples of linear regression between attribute scores of 0-10 and quartile mid points of attributes, to illustrate the process of rescaling. Points on the graphs represent attribute quartile midpoints (12.5, 37.7, 62.5 and 87.5 percentiles) regressed against scores of 2, 5, 7.5, 10. A) Stand basal area (ba): score= -3.7319 + 0.4352*(stand basal area) r^2= 0.99, p= 0.003 B) ln(Vegetation cover 0.5m-6m % +1) (logmedcover): score= 3.717 + 1.991* ln(% Vegetation cover 0.5-6m +1) r^2= 0.90, p= 0.052. The regression in b) is reduced in strength by the presence of numerous zero scores in the data.

3. Correlations between original and rescaled data

Very little information was lost in the rescaling process as the correlations between the original and rescaled data were all highly significant ($p<0.0001$) with Pearson's r values all being > 0.95.

4. Recalibrated index

The index scores derived for the 30 Tasmanian sites were normally distributed (Shapiro-Wilks test W = 0.97, p = 0.424) and the index successfully differentiated the study sites (Figure 2-4, Table 2-5). The index scores ranged from a minimum of 30.9% to a maximum of 84.8% with a mean of 59.5% which fell in between the McElhinny *et al.* (2006)'s dry sclerophyll forest index mean of 61.1% and woodland index mean of 58.5%.

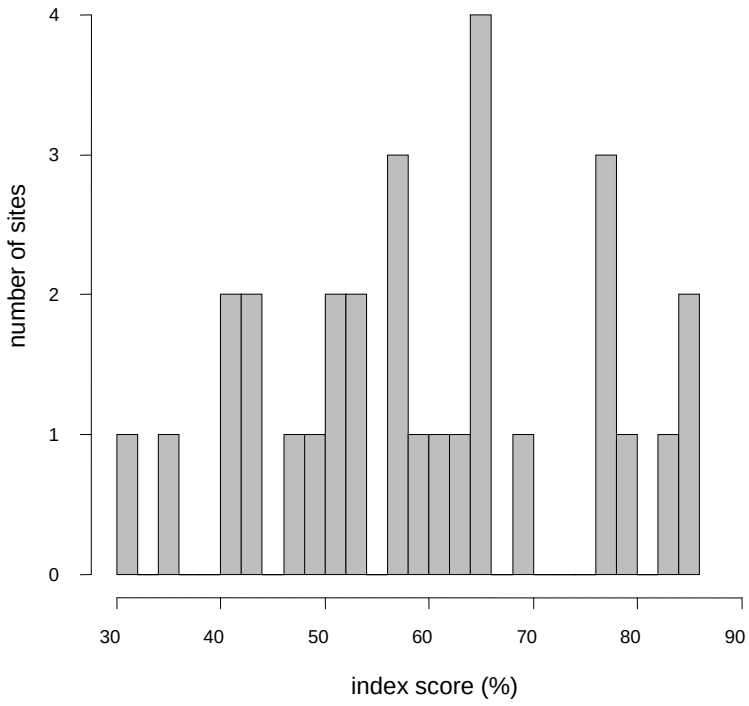

Figure 2-4 The recalibrated Tasmanian index differentiated between study sites and was normally distributed.

Table 2-5 The ranking and index score (%) of the 30 Tasmanian reference sites used to calibrate the McElhinny *et al.* (2006) structural complexity index. All but one healthy site were ranked within *a priori* remnant condition groups and sites from different locations and woodland types were distributed throughout the ranks.

Site	Class	Fire history	Location	Dominant	rank	score %
EC	Healthy	unburnt	Fingal	*E.amygdalina*	1	84.8
BOTH	Healthy	burnt 2005	Bothwell	*E.tenuiramis*	2	84.2
GR	Healthy	burnt 2002	Campania (S)	*E.tenuiramis*	3	83.8
WTG	Healthy	unburnt	Epping Forest	*E.amygdalina*	4	80.0
W1	Healthy	unburnt	Oatlands	*E.pauciflora*	5	78.0
TG	Healthy	burnt 2003	Epping Forest	*E.amygdalina*	6	77.1
LR	Healthy	unburnt	Campania (S)	*E.amygdalina*	7	76.9
B	Healthy	burnt 2004	Epping Forest	*E.amygdalina*	8	69.7
HH	Healthy	burnt 2002	Bothwell	*E.tenuiramis*	9	65.2
ED	Intermediate	unburnt	Fingal	*E.amygdalina*	10	64.8
DD	Intermediate	partially burnt	Epping Forest	*E.amygdalina*	11	64.6
ELD	Healthy	burnt 2006	Elderslie (S)	*E.tenuiramis*	12	64.6
HI	Intermediate	unburnt	Bothwell	*E.tenuiramis*	13	63.5
O	Intermediate	unburnt	Epping Forest	*E.amygdalina*	14	61.1
LS	Intermediate	unburnt	Campania (S)	*E.amygdalina*	15	59.4
MPT	Intermediate	unburnt	Oatlands	*E.pauciflora*	16	57.3
M	Intermediate	unburnt	Oatlands	*E.pauciflora*	17	56.8
W3	Intermediate	unburnt	Oatlands	*E.pauciflora*	18	56.3
BH	Intermediate	unburnt	Oatlands	*E.pauciflora*	19	53.5
CI	Intermediate	burnt 2005	Bothwell	*E.pauciflora*	20	52.3
EPT1	Poor	unburnt	Fingal	*E.amygdalina*	21	50.9
EPT2	Poor	unburnt	Fingal	*E.amygdalina*	22	50.2
DPT3	Poor	unburnt	Epping Forest	*E.amygdalina*	23	48.7
PT2W	Poor	unburnt	Oatlands	*E.pauciflora*	24	47.7
PT3W	Poor	unburnt	Oatlands	*E.pauciflora*	25	42.7
CPT1	Poor	burnt 2005	Bothwell	*E.tenuiramis*	26	42.2
HPT	Poor	unburnt	Bothwell	*E.tenuiramis*	27	40.6
DPT1	Poor	unburnt	Epping Forest	*E.amygdalina*	28	40.5
PT1W	Poor	unburnt	Oatlands	*E.pauciflora*	29	35.6
DPT2	Poor	unburnt	Epping Forest	*E.amygdalina*	30	30.9

The structural complexity index ranked the sites in a logical way with all but one of the *a priori* "healthy" sites ranked in the top 10. The exception was Elderslie Nature Reserve which had been burnt by intense wild fire two years prior to being measured. This site ranked at 12 out of 30, pushing the highest ranked "intermediate" site into the top 10 (Table 2-5). Sites from different locations and woodland community types were distributed throughout the ranks.

The index successfully distinguished between remnant conditions classes with a significant difference ($F_{2, 27} = 70.24$, $p<0.001$) between healthy (averaging a score of 76.4%), intermediate (59%) and poor sites (43%).

2.3.3 Ordination and evaluation of scored attributes

The ordination totally separated out the *a priori* remnant vegetation condition groups (Figure 2-5) with the exception of one intermediate site, the Downey Valleyfield remnant (DD) which had been partially burnt and had an unusually high amount of regeneration and medium vegetation cover for an intermediate site. The stress of the ordination, 0.104, is considered to represent a good fit (Clarke 1993).

Monte-Carlo analysis revealed that all attributes except Vegetation cover < 0.5m high were significant ($p<0.001$) vectors in explaining the variation among study sites (r^2 values are shown in Table 2-6). This shows that the 12 of the 13 core attributes chosen by McElhinny (2005) are also useful in distinguishing the respective structural complexities of remnant vegetation in Tasmania.

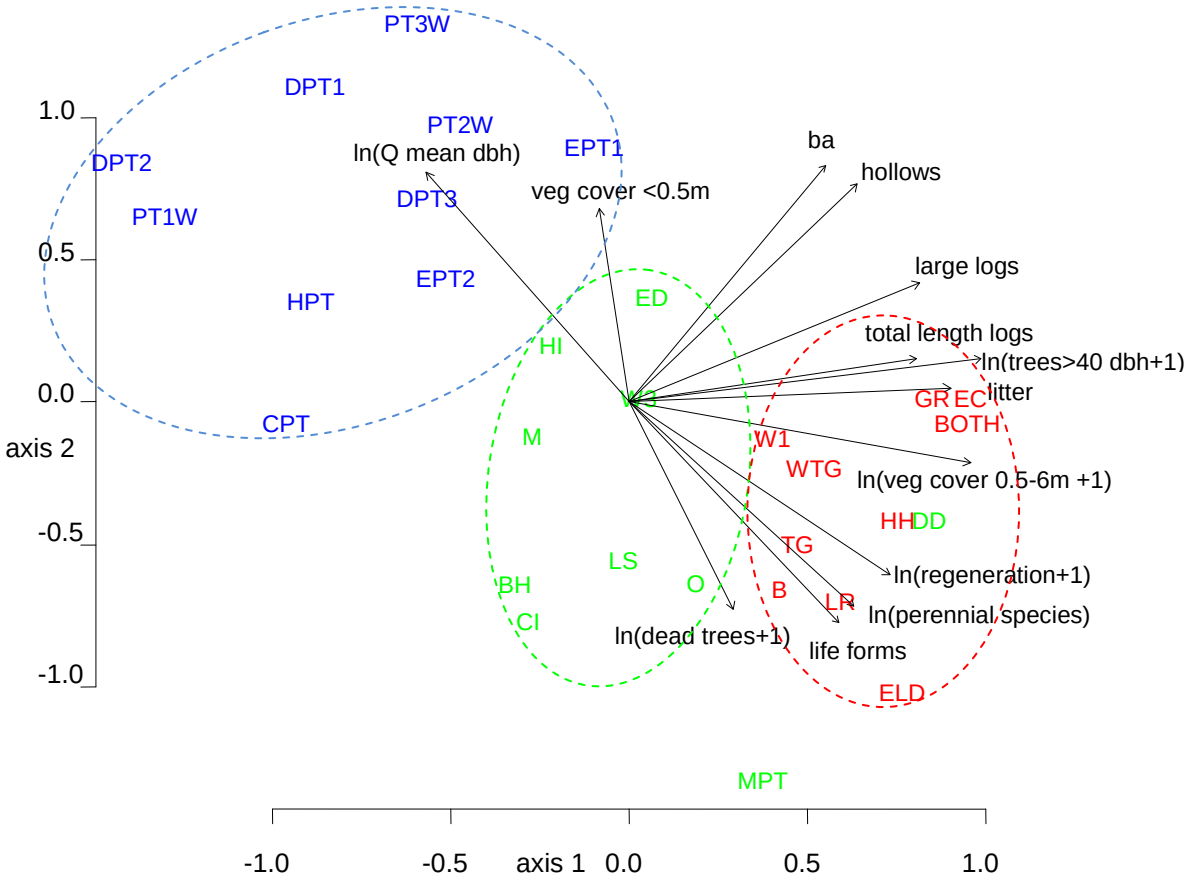

Figure 2-5 Semi strong hybrid multidimensional scaling of 30 Tasmanian sites (for site abbreviations see Table 2.1). The healthy sites are coloured red, intermediate sites are green and poor sites are blue. The 13 index attributes are represented by PCC vectors which display direction and magnitude of attribute variation among sites, in reduced dimensional space. Stress of the ordination is 0.104.

Table 2-6 The r^2 values for each structural attribute vector derived from multiple linear regressions in principal component correlation (PCC) analysis, describing how much of the variation of the attribute is accounted for by the vector. Significance of the vectors was tested using a Monte Carlo approach at p<0.01 level.

Attribute	r^2	significant PCC vector
Life form richness per 400m^2	0.864	*
ln(Perennial species richness per 400 m^2)	0.851	*
ln(Number of regenerating stems ha^{-1} +1)	0.844	*
ln(Quadratic mean dbh (cm))	0.824	*
Total log length (m ha^{-1})	0.689	*
Total large log length (m ha^{-1})	0.684	*
ln(Vegetation cover 0.5m-6m % +1)	0.650	*
ln(Number of dead trees ha^{-1} +1)	0.616	*
Litter dry weight (t ha^{-1})	0.517	*
Number of hollow bearing trees ha^{-1}	0.498	*
Basal area (m^2 ha^{-1})	0.465	*
ln(Number of live stems ha^{-1} >40cm dbh)	0.346	*
Vegetation cover < 0.5m (%)	0.183	ns

2.3.4 Location and community type

For all attributes, there were no significant differences among site locations. The only attribute that differed significantly among respective TASVEG groups was the total large log length where DAM (*E.amygdalina* on mudstone at Evercreech) had significantly more large logs than DPO (*E.pauciflora* on sandstone at Oatlands).

2.3.5 Correlations between regeneration and other structural attributes

The regeneration attribute, ln(number of regenerating stems ha^{-1} +1) was significantly and positively correlated with seven other structural attributes (Table 2-7) and negatively correlated with ln(quadratic mean of tree dbh).

Table 2-7 Correlations between number of ln (regenerating stems +1) and other measured structural attributes from 30 Tasmanian sites in order of strength.

Attribute	Pearson's r	p value
Life form richness per 400m^2	0.870	<0.001
ln(Quadratic mean dbh (cm))	- 0.866	<0.001
ln(Perennial species richness per 400 m^2)	0.850	<0.001
ln(% Vegetation cover 0.5m-6m +1)	0.767	<0.001
ln(Number of dead trees ha^{-1} +1)	0.677	<0.001
Litter dry weight (t ha^{-1})	0.613	<0.001
ln(Number of live stems ha^{-1} >40cm dbh)	0.480	0.007
Total log length (m ha^{-1})	0.458	0.011
Total large log length (m ha^{-1})	0.282	0.130
Vegetation cover < 0.5m (%)	- 0.277	0.137
Basal area (m^2 ha^{-1})	- 0.216	0.250
Number of hollow bearing trees ha^{-1}	0.020	0.916

2.3.6 Effect of remnant vegetation condition, fire and disturbance history

Eucalypt regeneration, defined here as at least one regenerating stem (Weinberg *et al.* 2011), occurred in 60% of the vegetation remnants studied. As was expected, there was significantly more regeneration in healthy remnants compared to remnants in intermediate and poor condition ($F_{2, 27} = 39.454$, $p<0.001$) and more in intermediate than poor sites (Figure 2.7A). The nine stands that had been burnt had significantly more regeneration than the twenty one unburnt stands ($F_{1, 28} =15.922$, $p<0.001$, Figure 2.7B) and in the analysis there was no apparent interaction between remnant health and fire history ($F_{2, 24} =0.669$, $p= 0.52$).

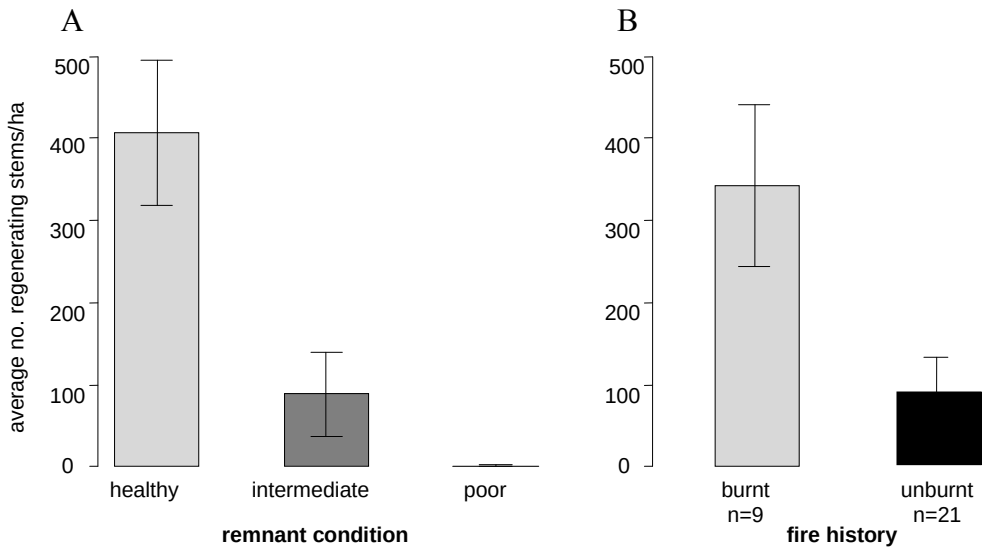

Figure 2-6 Average number of regenerating stems ha^{-1} (±1SE) in A) remnants of healthy, intermediate and poor condition, B) remnants that have or have not been burnt within the previous 10 years.

The amount of regeneration also varied significantly with disturbance history ($F_{3, 26} = 21.92$, $p<0.001$). The extremely disturbed sites had very few to no regenerating stems while the heavily disturbed sites had significantly more regeneration. The sites with moderate disturbance were not different in the number of regenerating stems from either heavily or lightly disturbed sites but lightly disturbed sites had significantly more regenerating stems than heavily disturbed (Figure 2-7). Only extremely disturbed (paddock tree) sites and one heavily disturbed stand did not have any regenerating stems. The one extremely disturbed paddock tree site that had a small number of regenerating stems had also been burnt.

The majority of regeneration observed was in the form of lignotuberous sprouts and saplings. True seedlings (recently germinated with minimal lignotuber development, a single stem, less than 50cm tall and usually no more than 8 pairs of expanded leaves) were only observed at four burnt healthy sites (Elderslie, Gravely Ridge, Tom Gibson and Barton), the burnt section of one intermediate site (Valleyfield) and

one unburnt healthy site (Radio Mast Hill Oatlands where two seedlings were observed).

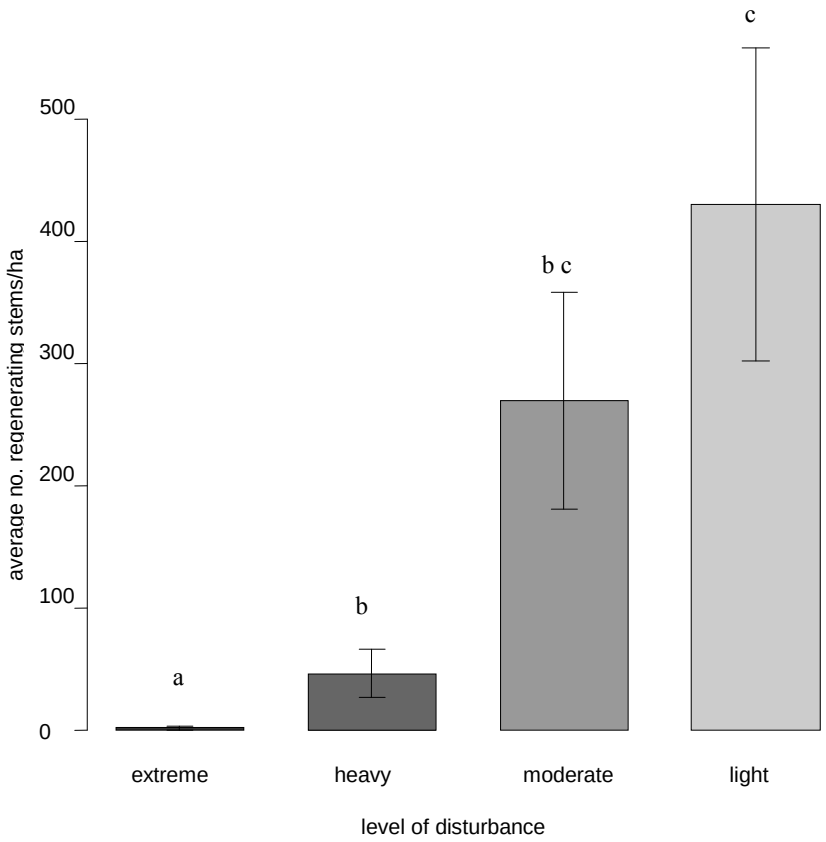

Figure 2-7 Average number of regenerating stems per hectare in sites of different levels of disturbance. Means that are significantly different ($p<0.05$) are indicated by different lower case letters.

Only a quarter of the stands had more than one regenerating stem per adult tree (Table 2-8).

Table 2-8 The minimum, maximum and average number of regenerating stems per adult (stem>5cm diameter) in remnants of healthy, intermediate and poor condition and the percentage of stands in each health class that have more than one regenerating stem per adult (ratio>1:1).

	Regenerating stems per adult			
Remnant Health Class	**Minimum**	**Max**	**Average**	**% of sites with ratio>1:1**
Healthy	0.11	4.59	2.21	60
Intermediate	0	1.39	0.44	20
Poor	0	0.8	0.08	0
Overall	0	4.59	0.91	26.7

The combined counts of tree diameter (dbh) in size classes for healthy sites (Figure 2-8) shows a classic reverse J-shaped curve which reveals that different size classes occupy the same area and thus the stands are well balanced between smaller and larger trees (Smith *et al.* 1997; George *et al.* 2005). The intermediate sites show an imbalance between smaller and larger trees revealing a lack of trees in smaller diameter classes and suggesting a long term (decadal scale) recruitment scarcity. With no stems less than 50cm dbh in poor sites, recruitment scarcity is even more pronounced.

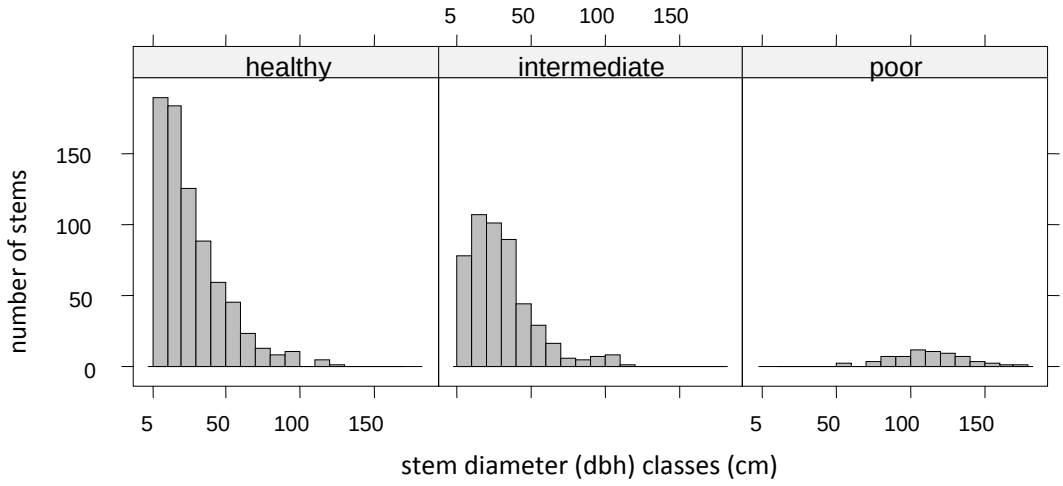

Figure 2-8 Tree diameter at breast height (dbh) distributions of stands in healthy, intermediate and poor condition classes. The first dbh class is actually 5-10cm as stems <5cm dbh were regarded as regeneration, all other class intervals are 10cm.

2.3.7 Remnant condition and structural attributes

Structural attributes varied among stands of different condition class (Table 2-9). Three attributes, basal area, vegetation cover <0.5m height and total large log length, did not significantly differ between any *a priori* condition classes. Four attributes did not differ significantly between healthy and intermediate sites only: life form richness, quadratic mean dbh, number of dead trees and litter dry weight.

Table 2-9 Attribute means of healthy, intermediate and poor sites with significant differences within each attribute indicated by different lower case letters. ns = not significant, * p <0.05, ** p < 0.01, *** p<0.001. (# original untransformed data)

	Condition class means[#]			
Attribute	**healthy**	**intermediate**	**poor**	**sig. diff.**
Perennial species richness per 400 m^2	19.48 [a]	13.83 [b]	5.53 [c]	***
Life form richness per 400m^2	7.88 [a]	7.26 [a]	4.23 [b]	***
Basal area (m^2 ha^{-1})	24.77 [a]	19.97 [a]	25.77 [a]	ns
Quadratic mean dbh (cm)	36.94 [a]	52.74 [a]	117.97 [b]	***
Vegetation cover < 0.5m height (%)	67.57 [a]	65.13 [a]	73.89 [a]	ns
Vegetation cover 0.5m-6m height (%)	20.41 [a]	5.18 [b]	0.03 [c]	***
Number of regenerating stems ha^{-1}	407.32 [a]	88.32 [b]	1.33 [c]	***
Number of hollow bearing trees ha^{-1}	21.66 [ac]	13.99 [b]	18.67 [cb]	*
Number of live stems ha^{-1} >40cm dbh	57.28 [a]	38.97 [ab]	24.67 [bc]	***
Number of dead trees ha^{-1}	54.33 [a]	53.01 [a]	8.66 [b]	***
Total log length (m ha^{-1})	1300.97 [a]	861.64 [b]	574.83 [b]	**
Total large log length (m ha^{-1})	227.65 [a]	155.17 [a]	134.02 [a]	ns
Litter dry weight (t ha^{-1})	13.90 [a]	10.92 [a]	6.24 [b]	*

2.4 Discussion

2.4.1 Regeneration and structural attributes

Regeneration is an important component of stand structure as the flux of seedlings to a juvenile state and then to a mature state determines the population structure of standing plants (Clarke 2000; Parkes *et al.* 2003; McElhinny *et al.* 2005). Regeneration also enables the stand to respond effectively to disturbance events and hence constitutes one of the more important drivers of forest/woodland dynamics (Florence 1981; Ashton 2000). In this study quantity of regeneration was significantly correlated with eight of the structural variables measured.

1. Life forms

The quantity of eucalypt regeneration was strongly correlated with the number of life forms present in a remnant stand. Regenerating stems were included as two separate life form categories (overstorey regeneration <2m height, overstorey regeneration >2m height) in the structural complexity index showing the importance of regeneration not only as a population process but also as a potential provider of physical structures for microhabitat in more than one vegetation stratum. A lack of regeneration, therefore, also results in a diminished range of physical structures in a stand.

2. Perennial species

The strong correlation between the quantity of eucalypt regeneration and the number of perennial species reflects the much greater amount of regeneration in healthier sites with a wide suite of species compared to the dearth of regeneration in paddock tree sites which had very few species. This supports the proposal by Yates and Hobbs (1997a) and Yates *et al.* (2000b) that degrading processes (such as fragmentation, grazing changed nutrient fluxes and invasion of exotic species) that reduce the diversity of native plant species in remnant vegetation also affect the presence of eucalypt regeneration. A reduced suite of perennial species also has implications for the diversity of eucalypt pollinator assemblages, which include a variety of insects and birds (Hingston and Mc Quillan 2000). Lack of year round nectar sources provided by multiple plant species may mean reduced pollinator diversity, pollinator populations, pollination services and seed set for eucalypts (Burkle and Alarcon 2011). A reduced complement of host plant species also decreases the diversity of mycorrhizal fungal communities and the inoculum levels of ectomycorrhizal fungi (Tommerup and Bougher 2000). The growth of eucalypts is inhibited in soils without

ectomycorrhizal inoculum (Ellis and Pennington 1992; Close and Davidson 2004) and lack of ectomycorrhizal fungi in degraded woodlands may play a role in limiting eucalypt establishment.

3. Quadratic mean dbh

The strong negative correlation between quadratic mean dbh and regeneration reflects the inclusion of paddock tree sites in the index as these stands had a few large trees with mostly no regeneration, while the healthier sites had more trees in a range of sizes and plentiful regeneration. This result is in agreement with Fischer *et al.* (2009) and Ottewell *et al.*(2010) who, working in dry agricultural areas of south-eastern and South Australia, found that scattered paddock tree sites had the greatest mean tree diameter with a symmetrical spread of diameters and no regeneration, while more intact remnant woodlands had smaller mean and median tree diameters, greater numbers of trees and an increased probability of regeneration.

4. Vegetation cover 0.5-6m high

The strong correlation between the quantity of eucalypt regeneration and the percentage cover of vegetation 0.5-6m in height reveals the importance of regeneration as a component of this vegetation stratum. The presence of eucalypt saplings adds to the physical distribution of foliage at this height along with other shrub and tree species. As the frequency of regeneration reduces (along with the number of large shrub species), the habitat resources provided by this stratum also declines. The richness of woodland bird species have been shown to be particularly sensitive to changes in this stratum of vegetation (MacDonald and Kirkpatrick 2003; Kutt and Martin 2010; Ford 2011; Munro *et al.* 2011).

5. Dead trees

The association between regeneration and the number of dead trees suggests evidence of gap phase regeneration i.e. regeneration occurring where there is a release from competition in the area around a tree following its death (Grubb 1977; Yates *et al.* 2000a). Tree death results in a break in the canopy, increasing the amount of light that reaches the forest floor with the likelihood of increased nutrient availability for individuals in the regeneration pool on the forest floor. Most importantly in dry woodland systems, tree death may also result in increased underground moisture availability for remaining stems (Stoneman *et al.* 1995; Yates *et al.* 2000a). Death of trees can occur not only as a result of episodic disturbance that results in recruitment events (such as fire) but also at other times, thus allowing some previously suppressed regeneration to access increased resources and be released from limiting competition to progress to the canopy as advanced growth (Florence 1996).

6. Litter

There was a significant positive correlation between the amount of litter and abundance of eucalypt regeneration which reflects the lower amount of litter near isolated paddock trees in pastures and in degraded remnants compared to healthy remnants which had higher stem densities. Generally, litter loads and types are highly variable spatially and temporally (Howell *et al.* 2006; McElhinny *et al.* 2010) and tend to be sparser in drier forests and woodlands compared to wet forests (Facelli and Pickett 1991; Facelli *et al.* 1999). The substantial litter loads in wet forests can have negative effects on germination and survival of seedlings through reduced light availability, being a mechanical barrier to emergence, damping off of seedlings by fungi and leaching of chemical germination inhibiters (Facelli and Pickett 1991).

However, in drier forests and woodlands where accumulation of litter is patchier and may not reach detrimental levels, litter may favour seedling establishment by reducing evaporative water loss from the soil, thus enhancing germination and/or seedling survivorship (Enright and Lamont 1989; Facelli and Pickett 1991; Howell *et al.* 2006). Litter patches may also suppress the emergence and growth of herbaceous and grass competitors (Facelli and Pickett 1991).

7. Trees >40cm dbh

Healthy sites had significantly more large trees and regeneration than poor sites and regeneration was positively correlated with the number of large trees. Woodland fragments containing few trees or a low density of paddock trees have a high likelihood of increased rates of inbreeding (Hardner *et al.* 1996) which impacts on seed set and offspring fitness (Burrows 2000; Mimura *et al.* 2009), reducing the likelihood of recruitment success. Poor seed set compounds recruitment problems (Turnbull *et al.* 2000). Trees within small woodland fragments may also set less seed because they are stressed. Isolated trees are more severely stressed physiologically than trees aggregated in remnant patches because edge trees are more frequently and more intensively exposed to environmental extremes (frost, drought, wind), soil compaction by stock, nutrient enrichment and attack by insects and mammalian herbivores (Landsberg and Wylie 1983; Davidson *et al.* 2007; Duncan *et al.* 2008) and may have low levels of ectomycorrhizal infection (Ellis and Pennington 1992; Close and Davidson 2004).

8. Total length of logs

The final attribute that was significantly correlated with quantity of regeneration was the total length of fallen logs. With aggregated length of logs up to 2km per hectare in healthy sites, the contribution of coarse woody debris (CWD) to the functioning of

dry woodland ecosystems should not be underestimated. Numerous studies have shown the importance of CWD in the provision of habitat for animals, birds, reptiles, insects, bryophytes and fungi (see Lindenmayer *et al.*(2002) for a review also (Yee *et al.* 2001; Mac Nally *et al.* 2002; Wardlaw *et al.* 2009; Brown *et al.* 2011) and the current study also implies a role of CWD in the regeneration process of the overstorey plant species. Studies overseas showed that CWD can protect seedlings from browsing animals (Milchunas and Noy-Meir 2002; Kupferschmid and Bugmann 2005; de Chantal and Granstrom 2007) while a local study has highlighted the role of logs as substrates for seedling germination in wet forests (McKenny and Kirkpatrick 1999). The role of CWD in the regeneration of eucalypts in woodlands will be explored further in the following chapters of this thesis.

The four structural attributes that were not correlated with abundance of eucalypt regeneration (length of large logs, % vegetation cover <0.5m height, stand basal area and number of hollow bearing trees), did not vary strongly along the gradient of degradation from healthy to poor sites as did the attributes that were correlated with regeneration abundance. While still adding to the ability of the structural complexity index to distinguish between individual sites, the length of large logs and stand basal area attributes did not significantly differ among condition groups (where regeneration abundance did). Cover of vegetation < 50cm high was also similar across all condition groups (although floristic composition was quite different) and was not useful in distinguishing between sites. There were less hollow bearing trees in intermediate sites than in healthy or paddock tree sites as a product of tree size, maturity and age. The number of hollow bearing trees per hectare did therefore not correlate with the number of regenerating stems.

2.4.2 Effect of remnant vegetation condition, fire and disturbance history on regeneration

In dry woodlands, episodic recruitment events are " stored" (Warner and Chesson 1985) in regeneration pools that persist over time in healthy stands (Duncan 1999). The results of this study suggest that these pools of seedlings and lignotuberous sprouts are degraded or diminished in parallel with the decline in overall condition of remnant vegetation. All ten sites identified in this study as healthy had significantly greater numbers of regenerating stems than intermediate sites and poor sites, with intermediate sites having less than one quarter of the regenerating stems observed in healthy sites and paddock tree stands having none. This is consistent with other studies (Gibbons *et al.* 2008a; Fischer *et al.* 2009; Weinberg *et al.* 2011). For example Gibbons *et al.* (2008a) showed that, in the south-eastern Australian temperate grazing region, 97% of 462 (x 0.1ha) reference plots in relatively undisturbed dry schlerophyll woodland and forest contained eucalypt regeneration (defined as stems <5cm diameter) while Dorrough and Moxham (2005) only found eucalypt regeneration in 27% of 519 dry grassy forest and woodland sites in Victoria which had a range of disturbance intensities.

While 60% of sites surveyed in the current study supported regenerating stems in the form of lignotuberous sprouts and saplings, only six sites (20% of all sites) showed evidence of recent recruitment in the form of new seedlings (defined as single stems with minimal lignotuber development). All except one of the sites exhibiting new seedling recruitment had been burnt in the previous six years. The number of regenerating stems was also significantly greater in sites that had been burnt within the last ten years compared to sites that had not been recently burnt. This highlights the important role of fire in the recruitment of eucalypts in dry woodland remnants

and supports suggestions that recruitment of new individuals into the regeneration pool is mainly dependent on the episodic disturbance of fire (Yates *et al.* 1996; Gill 1997). Indeed lack of fire may be regarded as an important abiotic barrier to recruitment that maintains sites in a non regenerating state (Suding *et al.* 2004).

There is a strong link between the health of woodland remnants, the presence of regeneration and management regimes, particularly livestock grazing (Yates *et al.* 2000b; Davidson *et al.* 2007; Fischer *et al.* 2009). In this study 60% of sites contained eucalypt regeneration with the majority of regeneration occurring in sites that were only lightly to moderately disturbed by grazing, with heavily grazed sites showing very restricted amounts of regeneration and extremely disturbed sites mainly none. This result is very similar to a recent study on the extent and pattern of eucalypt regeneration in an agricultural region of the Murray-Darling Basin in south-eastern Australia (Weinberg *et al.* 2011). Of 149 sites measured there, 58% of samples contained regeneration, with grazing intensity being the most influential variable affecting its presence or absence. Similarly, Fischer et al (2009) also working in south-eastern Australia, found that remnant vegetation sites with no grazing and fast rotation (light) grazing were much more likely to support eucalypt regeneration than sites that endured slow rotation (heavy) and continuous (extreme) grazing.

Sites that had a history of light or moderate grazing (i.e. < 2DSE) did not have significantly different quantities of regeneration in our study and in the Weinberg *et al.* (2011) study the probability of regeneration in sites that were lightly to moderately grazed was the same. Dorrough and Moxham (2005) also reported the presence of eucalypt regeneration in sites subject to intermittent grazing. This suggests that eucalypt regeneration is not completely incompatible with intermittent

livestock grazing up to a level of moderate intensity, however extreme intensity of grazing may cause a threshold to be passed that almost completely prohibits regeneration (Spooner and Allcock 2006). Perennial species richness has also been shown to be directly affected by intensity of grazing (Gilfedder and Kirkpatrick 1998; Yates *et al.* 2000b; Leonard and Kirkpatrick 2004; Kirkpatrick *et al.* 2005; Dorrough *et al.* 2006; Dorrough and Scroggie 2008) with Dorrough *et al.* 2006 finding that intermittent livestock grazing was associated with higher native plant species richness than either continuous or no grazing but only at low soil phosphorus.

Three of the ten sites rated highest in the structural complexity index in our study (Humbie Hill, Barton, Lowdina Regrowth) had past histories of moderate levels of grazing. This shows that sites previously exposed to moderate levels of disturbance can have high levels of structural complexity, in agreement with the findings of McElhinny *et al.* (2006). Moderate levels of disturbance do not necessarily exclude, and may even encourage, regeneration, species richness and good habitat (Connel 1978). Indeed it has been shown that some rare and endangered species in the Midlands require disturbance to flourish (Kirkpatrick and Gilfedder 1995). Kirkpatrick et al (2005) suggest that remnants need a range of management prescriptions in order to maintain a diversity of species across the landscape.

In contrast, sites of intermediate and poor health in this study had histories of high to extreme grazing pressure and were depauperate in perennial species, eucalypt regeneration and medium level (shrubby) vegetation cover. This suggests that ecological barriers (or thresholds) may have been passed at these sites (Spooner and Allcock 2006). Change in management regimes and further intervention to reverse these outcomes is required in order to facilitate regeneration of dominant eucalypts and understorey in these woodlands and force a transition back to a healthier state

(Hobbs and Norton 1996). If no efforts are made to restore regeneration, many of these sites could become bereft of trees in decades to come (Gibbons *et al.* 2008b; Fischer *et al.* 2009).

2.4.3 How much is enough eucalypt regeneration?

Neyland (1996) suggests that in Tasmanian woodlands affected by tree decline, successful regeneration has occurred when there are at least 200 seedlings per hectare distributed such that there is at least one seedling for every $50m^2$ of the remnant area. Subsequently, there should be at least 50 regenerating stems (sprouts or saplings <5 cm diameter) established per hectare. Thirteen out of the 30 sites in our study had more than 50 regenerating stems per hectare.

Forestry technical manuals focus on post-harvest stocking rates for successful regeneration and suggest that almost a thousand (976) seedlings, 156 large saplings (approximately 15cm dbh), and 83 adult trees (>25cm dbh) per hectare (equivalent to 239 stems>5cm dbh per hectare) represents an adequately stocked regenerating dry forest (Orr 1991). This translates to a regenerating stem to adult ratio of 4:1. Using these constraints, only one site in our study, the most recently burnt site Elderslie, has anywhere near adequate regeneration (with 963 regenerating stems per hectare) and only seven sites (not including Elderslie) are adequately stocked with adults (stems >5cm). However these figures apply to sites that are healthy native dry forest which have recently been harvested and reseeded with a focus on future advance growth for timber production. Additionally, they are more applicable to production forests rather than woodland densities and most sites in our study would be classified as woodlands.

In a demographic study of eucalypts in the Mount Lofty bioregion of South Australia Ottewell *et al.* (2010) found that the average number of regenerating stems per adult tree surveyed in relatively healthy woodland remnants was 2.7 ± 0.56 (SE) for *E. camaldulensis* and 3.3 ± 0.83 for *E. leucoxylon* and that stem diameter (dbh) size classes were strongly skewed to smaller sizes. Only five of the sites in our study had a regeneration to adult ratio greater than 2:1. Of the eight sites whose ratio exceeded 1:1, six had been burnt within the past ten years and thus the ratio reflects relatively recent recruitment events.

Population maintenance at any site requires the presence of each growth stage (senescing old tree, mature tree, sapling, seedling/sprout) in increasing numbers from mature old adult tree (few) to seedling (many) (George *et al.* 2005). Zavaleta *et al.* (2007) suggest that a low sapling to adult ratio (i.e. < 1:1) indicates long-term (decadal scale) recruitment scarcity in Californian Oaks. In the current study, lack of trees in small size classes in both intermediate and poor sites relative to healthy reveal similar long-term recruitment scarcity in Midlands' eucalypts and an unavoidable decline in future availability of mature trees (Vesk and Mac Nally 2006; Gibbons *et al.* 2008b; Fischer *et al.* 2009). Modelling by Gibbons *et al.* (2008b) suggested that a minimum of two regenerating stems per adult (in conjunction with minimal mortality of mature trees) was needed to ensure long term maintenance of scattered tree cover in dry agricultural districts. Nearly three quarters of study sites in the current study had a regenerating stem to adult ratio <1:1 suggesting that regeneration pools (the majority of which are made up of saplings and lignotuberous sprouts) are less than is necessary for long term population replacement.

2.4.4 Index calibration

Combining a range of dry eucalypt communities from a broad range of locations across the Midlands into one structural complexity index was a reasonable decision as all the structural attributes, except one, did not differ between locations or forest types. The attribute that differed (length of large logs) was only significantly different for one pair of the six TASVEG* vegetation communities. This difference was between a) Evercreech (DAM*) where artificially high numbers of logs were present in piles created from wood outside the stand (pushed up by bulldozer) during adjacent plantation establishment and b) Oatlands (DPO*) where woodland remnants were artificially low in numbers of logs because the surrounding farmland had been heavily exploited for firewood.

Calibrating the McElhinny index to Tasmanian conditions resulted in a logical ranking of sites with nine out of ten of the "healthy" sites ranked in the top 10 and all the "poor" sites ranking in the bottom 10. With all but one attribute vector (vegetation cover <0.5m height) being significant in separating out sites in the ordination, and having the three subjectively identified condition groups clustered separately, it can be concluded that measuring this relatively limited set of structural complexity attributes has successfully captured the range of condition in the measured sites. This strongly supports the selection of variables by McElhinny *et al.* (2006) and the wider application of this approach beyond their original samples.

The ranges and means of the Tasmanian attribute data were similar to those of the South eastern Highlands Bioregion (SHB) studied by McElhinny (2005) for seven out of the thirteen attributes, while values for vegetation cover, log lengths and tree size were generally greater in Tasmania. These differences highlight the necessity of recalibrating the index so that each attribute is ranked within the range of variation

shown across the particular region of interest (Gibbons and Freudenberger 2006; McElhinny *et al.* 2006). If the McElhinny index had not been recalibrated for Tasmanian conditions it would have given maximum scores for the vegetation cover, log length and large tree attributes, reducing the ability of the index to distinguish between sites.

Twelve out of thirteen of the structural attributes chosen by McElhinny (2005) for his index of structural complexity for the SHB were also able to distinguish between sites in the woodlands of the Midlands of Tasmania. This indicates that if the full process undertaken by McElhinny to arrive at the core attributes (i.e. measurement and redundancy analysis of 70 structural variables) were to be undertaken in Tasmania it is quite likely that a very similar suite of attributes would have been derived.

The two vegetation cover attributes were problematic in the calibration process, with the low cover (<0.5m height) attribute being the one that did not significantly distinguish between sites, and the medium cover attribute (0.5-6m height) not having a significant r^2 when quartiles were regressed against scores. Both attributes lacked a normal distribution. This was caused by an outlier in the case of the low vegetation cover and the presence of numerous zero scores in the medium vegetation cover. Both outliers and numerous zero scores are common in biological data and should be kept in an analysis when they are true measurements despite making statistical analysis more difficult (Quinn and Keough 2002). Further, levels of vegetation cover have been included in the majority of habitat complexity assessment schemes (including McElhinny *et al.* 2006) because vegetation cover has been shown to be an important element in the provision of habitat for birds, ground dwelling mammals and insects (e.g. (Catling and Burt 1995; Watson *et al.* 2001; MacDonald and

Kirkpatrick 2003; Parkes *et al.* 2003). These attributes have therefore also been retained in this current index, despite the difficulties described above.

The calibrated Tasmanian index now represents a tool that can be used to assess the structural complexity of other similar assemblages of remnant vegetation and revegetation across the Midlands of Tasmania relative to the variation measured in the region. It also provides an indication of the natural range of attributes in healthy sites towards which restoration efforts in degraded sites could aim. In addition, this methodology could be used to monitor sites over time and to assess the effectiveness of any restoration treatments or change in management regime (Hobbs and Norton 1996).

The advice given by McElhinny (2005) to encourage regeneration is to"create a receptive seedbed; apply a cool burn; expose patches of bare mineral soil", which is similar to advice given in forestry technical manuals on regenerating dry forests (McCormick 1991; Orr 1991; Orr and Todd 1992; ForestryCommission 1993; Neyland 1996). However these manuals also suggest that achieving eucalypt regeneration can be difficult, particularly on sites with grassy understoreys and/or those that have been grazed. Numerous other studies have shown that establishing eucalypt regeneration in grassy understorey and grazed land is extremely difficult (Ellis and Pennington 1992; Fensham and Kirkpatrick 1992; Pinkard 1992; Clarke and Davidson 2001; Clarke 2002; Li *et al.* 2003; Semple and Koen 2003; Skinner *et al.* 2009; Orscheg *et al.* 2011). This suggests that further research into the regeneration niche of eucalypts in dry agricultural areas is warranted.

Hobbs and Cramer (2003) recommend that ecosystem processes in eucalypt woodland systems need to be examined at a range of spatial scales. In this chapter,

the processes of eucalypt regeneration have been examined on a stand scale by assessing the quantity of regeneration in remnants of varying condition. The next chapter focuses on investigating finer (microsite) scale elements of the eucalypt regeneration process in healthy woodland remnants which may then inform restoration and assist in the development of new techniques of establishing eucalypt regeneration in degraded remnants within dry production landscapes.

Chapter 3 Microsite attributes and eucalypt regeneration

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Chapter 4 Soil water repellency and eucalypt regeneration

4.1 Introduction

Soil water repellency occurs when the cohesive forces between water molecules are stronger than the adhesive forces between water molecules and dry soil particles and thus a water drop placed on the soil surface will stay as a spherical drop on the soil surface rather than being absorbed into the soil (Figure 4-1). The repellence is attributed to hydrophobic organic compounds found in interstitial spaces and coating soil particles and aggregates (Doerr *et al.* 2000). These compounds originate from decomposing organic matter, plant leaf waxes and root exudates as well as from fungal and microbial activity. A range of compounds have been implicated including aliphatic hydrocarbons, fatty acids and waxes (Hallett and Young 1999; Doerr *et al.* 2000; Franco *et al.* 2000; Atanassova and Doerr 2010).

Figure 4-1 Water droplets sitting on the soil surface indicate soil water repellency in a eucalypt forest in Epping Forest, Northern Midlands Tasmania.

Severe levels of soil water repellency naturally occur in dry eucalypt forests in Australia (McGhie and Posner 1980; Crockford *et al.* 1991; Doerr *et al.* 2004; Doerr *et al.* 2006; Howell *et al.* 2006; Shakesby *et al.* 2007), where eucalypts are grown in dry land plantations elsewhere in the world (Doerr 1998; Scott 2000; Keizer *et al.*

2005; Leighton-Boyce *et al.* 2005) and now in Tasmanian woodland remnants (Chapter 3, Figure 4-1). Fire affects water repellency in these soils such that repellency is intensified by soil temperatures up to 260 °C and is abruptly removed at specific temperatures between 260 and 340 °C with the temperature threshold for repellency destruction decreasing with increased heating duration (Doerr *et al.* 2004). Due to steep temperature gradients in dry soil under intense fire these effects can result in a wettable surface layer of soil being undelayed by a severely hydrophobic layer (DeBano 2000; Letey 2001; Doerr *et al.* 2006).

The majority of studies of water repellency in soils supporting eucalypts have focused on the hydrological and geomorphological implications of soil water repellency after fire, particularly the effects on post fire erosion. While a number of authors suggest that there would also be implications for seedling germination and establishment, only one study (Howell *et al.* 2006), has looked specifically at the possible effects of water repellency on the eucalypt forest floor as a seed bed. Howell *et al.* (2006) focused mainly on the distribution of water repellency in relation to surface litter and microtopographic features before and after a low intensity fire. While they found that water repellency was not directly related to the presence or absence of surface litter, they concluded that "variable water repellency at the soil surface and within the profile may contribute to differential survival of seedlings after fire".

Inherent water repellency in eucalypt woodland soils promoted by hydrophobic substances from adult trees could play a large role in the restriction of eucalypt recruitment in inter fire periods. Poor germination on hydrophobic soils has been reported for annual crop and grass species such as barley (Bond 1972), Wimmera ryegrass (Osborn *et al.* 1967), crested wheatgrass and blue bunch wheatgrass

(Madsen 2010). Aspects of soil water repellency that may influence seedling germination and establishment include its effect on water infiltration, movement and storage in the soil (Osborn *et al.* 1967; Dekker and Ritsema 1994; Doerr *et al.* 2000; Doerr and Moody 2004; Howell *et al.* 2006), its variability through space and time (Doerr and Moody 2004), and its ability to disconnect seeds and seedlings from soil moisture (Madsen 2010).

The spatial variability of water repellency in soils has been shown to affect the way water infiltrates, moves through and is stored in the soil profile (Ritsema and Dekker 1996; Doerr *et al.* 2000; Lamparter *et al.* 2006; Nyman *et al.* 2010). Water repellency at the soil surface reduces infiltration capacity (Wallis *et al.* 1990; Doerr *et al.* 2003) and can cause runoff and overland flow especially after prolonged dry periods and after fire (McGhie and Posner 1980; Crockford *et al.* 1991). The extent of the overland flow is determined also by the frequency of pathways through the hydrophobic layer such as macropores (inter-aggregate pore spaces, shrinkage cracks, root channels), rocks, disturbances from animal and ant activity and patches of wettable or less repellent soil (Doerr *et al.* 2003; Eldridge and Freudenberger 2005; Shakesby *et al.* 2007; Urbanek and Shakesby 2009; Nyman *et al.* 2010). Where slight differences in microtopography or surface hydrophobicity occur rainwater can pond and the hydrostatic pressure of the head of water may eventually break through the water-repellent layer and create isolated, vertically directed flow paths deeper in the profile. The movement of water through these pathways can lead to what has been called “preferential flow” or “fingered flow” where water bypasses the soil matrix via preferred pathways (Ritsema and Dekker 1996).

Preferential flow can cause the soil between conducting pathways to remain dry even after heavy rainfall events or even throughout an entire wet season (Dekker and

Ritsema 2000; Doerr *et al.* 2000). Dekker and Ritsema (2000) have shown that centimetre-scale spatial variations in repellency can cause the formation of preferential flow pathways and that water content of adjacent soils can vary widely. Preferential flow can also lead to higher moisture in the subsoil as water bypasses the dry soil bodies and rapidly moves through to lower layers (Imeson *et al.* 1992; Ritsema and Dekker 1996). Robinson *et al.* (2010) working in a pinion pine/juniper woodland in the USA, suggested that these trees "engineer" their environment by creating a water repellent surface that concentrates water below the surface through preferential flow giving them an ecohydrological advantage over shallow rooted understorey species.

Where a water repellent layer is overlain by a wettable layer (which may result from localised intense fire) rain water tends to pond above the hydrophobic layer and may be directed laterally to adjacent channels of preferential flow. This has been called "distribution flow" (Dekker and Ritsema 1995; Doerr *et al.* 2000). Moisture may also be stored in the top wettable layer. This moisture may facilitate germination of seeds, may be used for transpiration by plants with roots in this zone or be evaporated from the soil surface (Doerr *et al.* 2000; Shakesby *et al.* 2007).

The moisture stored in an overlying hydrophilic layer may eventually cause a phase change in the underlying hydrophobic layer from repellent to wettable. This may be the case after prolonged wet conditions as water repellency has been shown to be temporally as well as spatially variable (Crockford *et al.* 1991; Doerr and Moody 2004; Leighton-Boyce *et al.* 2005; Keizer *et al.* 2008; Lemmnitz *et al.* 2008; Jordan *et al.* 2010). Crockford *et al.* (1991) periodically measured the water repellency of soils in a dry eucalypt forest over four years and found that a number of weeks of consistent wet weather was needed to break down severe hydrophobicity and a much

greater than normal frequency of rain to keep the soil wettable. After an extended period of wettability the soil returned to its previously repellent state after only a week of dry weather. Such change in soil repellency is usually related to the changes in the moisture content of the soil as the soil wets up and dries out (Dekker and Ritsema 1994; Leighton-Boyce *et al.* 2005).

The variability of soil water repellency over time and with soil water content has lead to a differentiation between "actual" and "potential" repellency (Dekker and Ritsema 1994). This has implications for measurement of soil water repellency as measurements made in the field or on field moist samples are measurements of the "actual" water repellency at that time, while measurements made on soils that are sampled and dried in the laboratory are of their "potential" or inherent water repellency.

This variability over time also has implications for the germination and survival of seedlings. Seeds that fall onto soils that are actually severely water repellent may not have adequate water to germinate at all as water repellency can promote runoff, reduce infiltration and disconnect seeds from underlying soil moisture reserves (Osborn *et al.* 1967). Seeds that fall into temporarily wettable soils with a high underlying potential repellency may germinate but subsequently become cut off from soil moisture reserves as the soil dries out and water repellency becomes re-established (Madsen 2010). Indeed soils with severe potential water repellency may represent a barrier to seedlings making the transition from germination to establishment in eucalypt woodland soils.

In Chapter 3 of this thesis the majority of seedlings surveyed had established in ashbeds (95 +/- 3.2 %) and 80 ±4.7% were sheltered by coarse woody debris. The

majority of seedling microsites surveyed had wettable to moderately water repellent soils while the majority of microsites without seedlings had moderate to extreme soil water repellency. Therefore, the water repellency characteristics of the typical seedling microsite and its surroundings warranted further investigation and will be the focus of this chapter.

The three research questions of this study were:

1. What are the soil moisture and water repellency characteristics of microsites on the forest floor both adjacent to and away from coarse woody debris A) where fire has occurred in the last 5 years and B) where forest has been unburnt for at least 10 years?
2. In ashbeds extending from coarse woody debris, how does soil water repellency vary horizontally (with distance from log) and vertically (with soil depth)?
3. What are the implications for eucalypt recruitment?

4.2 Methods

4.2.1 Water repellency tests

While soil water repellency has increasingly been studied throughout the world, with an exponential increase in publications since 1960 and averaging 200 related articles every five years (Dekker *et al.* 2005b), the methodology for its assessment has not been consistent (Dekker *et al.* 2009). Numerous methods of measuring and classifying soil water repellency have been developed and most of these are described by Hallett and Young (1999), Letey *et al.* (2000), Roy and McGill (2002) and Ramirez-Flores *et al.* (2010).

Two of the most commonly used methods to asses soil water repellency are the water drop penetration time (WDPT) test and the molarity of ethanol droplet (MED) test (Letey *et al.* 2000). The WDPT test determines whether a soil is repellent or not and how persistent the water repellency is while the MED test measures the severity of water repellency. Both of these tests are easy and inexpensive to employ and are useful for rapid determination of water repellency both in the field and in the laboratory (Jordan *et al.* 2010) and were used in this and the previous study.

The methods used for the WDPT and MED tests were the same as those described in section 3.2.4 in Chapter 3 of this thesis. Each sample was allocated a water repellency class as described in Table 3.2.

4.2.2 Sites

Liliesleaf Farm Oatlands

"Liliesleaf" is a sheep farm 5km south east of Oatlands in the Southern Midlands of Tasmania on which a hundred hectares of eucalypt woodland of *Eucalyptus viminalis* and *E. pauciflora* (42.32°S 147.40°E 454m elevation) has been covenanted under the private forest reserve system and fenced to exclude livestock. Approximately a third of the reserve, on a west facing sandstone hillside, was accidentally burnt in December 2007. The fire was generally of moderate intensity (estimated scorch height of about 4m) but many of the larger logs were mostly consumed by the fire, leaving obvious ash beds and large charcoal covered sections of burnt log. A study of the water stored under logs in the burnt area and in adjacent unburnt forest was conducted eleven months later in November 2008 (Experiment 1). A total of 363mm of rain had fallen in that eleven month period at the Oatlands Post Office meteorological station, 3.2km from Liliesleaf. In the previous two years 2006-2008,

the farm experienced well below the long term (since 1882) average annual rainfall of 552.9mm (BOM 2011)

Forton Farm Epping Forest

"Forton" at Epping Forest in the Northern Midlands of Tasmanian (41.72°S 147.32°E 170m elevation) experienced a wildfire on 22nd January 2009 that burnt through paddocks and a number of remnant patches of *Eucalyptus amygdalina* open forest. In one 3 hectare remnant there was a range of fire intensity, including high intensity (scorching into the upper canopy of the trees), low intensity with some hot fires around logs, and patches of unburnt forest. Following the fire large amounts of partially burnt coarse woody debris was present on the forest floor (Figure 4-2).

Figure 4-2 An intensely burnt section of a *Eucalyptus amygdalina* forest remnant at Forton Farm Epping Forest 2 months after fire. The log in the foreground was sampled in Experiment 2.

Forton had experienced three years of below average rains prior to the fire but above average rainfall (586.9 mm) occurred in 2009 following the fire. The long term average at Forton, which has its own meteorological station, is 563.1mm per annum

(BOM 2011). A site inspection and pilot study was undertaken at this site in April 2009 when conditions were dry (Experiment 2) and more intensive water repellency measurements were made in August 2009 when conditions were wet (Experiment 3).

4.2.3 Experiment 1: Liliesleaf Farm Oatlands

The aim of the sampling at Oatlands was to characterise the water content of soil under and away from burnt and unburnt logs. The amount of water present in the surface soil (0-10 cm) was assessed beneath logs >50 cm in diameter and 5m away on the forest floor at two locations: burnt forest and adjacent unburnt forest, at a fire boundary on the west facing slopes of the reserve. In the burnt area logs were partially burnt and in the unburnt area logs were intact, but were partially decayed where they contacted the ground.

At each log, for 5 burnt and 5 unburnt logs, soil samples were taken at 5 points beneath the log where it contacted the ground. Samples were taken by either rolling the log over or digging under the log. For each log, a position 5m from the log was subjectively chosen and 5 surface soil samples were taken.

All samples were enclosed in plastic zip-lock bags and then placed in a cooler. The moist soil samples were weighed the next day, oven dried at 105°C for 48hrs and then reweighed. The relative soil moisture content was then expressed as a percentage of the dry weight of the soil.

4.2.4 Experiment 2: Forton Farm Epping Forest

During the pilot study 3 months after fire, no eucalypt seedlings were observed. The aim of the pilot study at Epping Forest was to investigate a number of sampling techniques in order to refine methods for the subsequent replicated study. The initial

sampling also allowed the soil at the site to be characterised down to a depth of 80cm.

Two large partially burnt logs were chosen that enclosed microsites typical of those found to support eucalypt seedlings in the previous study reported in Chapter 3. Soil samples were taken next to the two logs and 5m away from each log to characterise the soil near and away from logs. This methodology was replicated for a single log and away from log pair in an adjacent unburnt area in order to characterise the pre fire soil condition. In the unburnt area microsites near logs had no ashbed and were covered in grass and fine litter.

Sampling was conducted by digging trenches. Soil samples were taken at depth 0-2.5, 2.5-5, 5-7.5, 7.5-10, 10-15, 15-20, 20-30 cm at all trenches but also at 30-40, 40-50, 50-60, 60-70 and 70-80cm depths at near the first log sampled (log 1). The samples collected from each trench at each depth were placed in separate labelled plastic bags and placed in a cooler.

In the laboratory, the field moist soil samples were individually weighed then placed in a 100°C oven for 7 days and reweighed to enable relative water content calculations. Both WDPT and MED tests were undertaken (as described in section 3.2.4) on subsamples (approximately 10g per subsample) that had been left to equilibrate to room temperature for 24 hours. A Spearman's rank correlation test was undertaken on data from the MED and WDPT tests on dry samples to ascertain whether it was necessary to undertake both tests in further studies.

Results from the tests on these subsamples suggested that there was considerable variability in water repellency in the top 5cm of soil at a finer scale than that sampled. Subsequently a representative log at Forton Farm was selected for *in situ*

finer scale water repellency tests. Only MED tests were undertaken in the field as it was more time efficient than the WDPT test and the data from the previous tests were strongly correlated.

Figure 4-3 Soils were excavated in 1cm layers and MED tests undertaken on each surface. Intact water drops (0% ethanol) on the surface 2cm deep in an ashbed show a water repellent layer

The log chosen at Forton Farm ran NNE – SSW and had its central section burnt through. A transect was run NW of the log through the strongest gradient of burn intensity from high to low. Six points along the transect were sampled for water repellency *in situ*: in the ashbed where the log was burnt through then 30, 60, 90, 120 and 200cm from the first point. At each point, MED tests were undertaken on the surface soil then at 1 cm intervals down to 5cm depth. The soil was excavated using metal rulers such that 1cm of soil was removed at a time and a flat surface remained (Figure 4-3).

4.2.5 Experiment 3: Forton Farm Epping Forest

After the pilot study was conducted, it was decided to wait until eucalypt seed germinated so that soil could be sampled near seedlings. Thus the main round of sampling was done in August after good winter rains and prolific eucalypt germination. The soil was too wet to undertake *in situ* testing so the samples were

taken back to the laboratory and tests were undertaken to determine “potential” water repellency (Dekker and Ritsema 1994).

Soils were sampled in eight ashbeds at five distances from the central log in each ashbed. The first sample at each log was taken either directly under an overhanging part or directly adjacent to the log and within 10cm of a newly germinated eucalypt seedling. The other samples were 20cm, 50cm, 80cm and 110cm away from the original point near log. There were instances where sampling points landed near to smaller pieces of partially burnt branches. No points were sampled past 110cm as often a new ashbed around a different log was reached within 2 metres due to the large amount of woody debris on the ground.

At each distance from log, 6cm x 6cm squares of soil were excavated using metal rulers in centimetre depths down to 5cm. Each 6 x 6 x 1cm sample was placed in a separate plastic bag, sealed, labelled and placed in a cooler for storage and transportation to the laboratory. Distance to the nearest seedling was measured at each sampling point.

The first log was sampled on 7/08/2009. The samples were weighed, air dried for 3 days at room temperature and then reweighed to gain a measurement of relative water content (RWC). Water repellency measurements were made and then the samples were put in an oven at 100 °C for 48 hours. They were then reweighed and remeasured. The differences in RWC and water repellency measurements in the air dried and oven dried samples were not significant, thus the samples from the rest of the logs were only air dried. The remaining seven logs were sampled over the next fortnight, with all samples being air dried under laboratory conditions for 3-4 days prior to repellency measurements being taken.

4.2.6 Data analysis

All statistical analyses were undertaken using the statistics package R version 2.9.0 (R Development Core Team 2009). MED and WDPT classes were not normally distributed and can be regarded as categorical variables. Thus only non-parametric tests were undertaken on this data (Scott 2000; Quinn and Keough 2002).

Experiment 1

Analysis of variance was undertaken on relative water content (RWC) data to compare the effect of log and the effect of a previous fire on soil surface moisture.

Experiment 2

A Spearman's rank correlation test was undertaken on MED and WDPT data to ascertain whether it was necessary to undertake both tests in further studies. Spearman's rank test checks for monotonic relationships between variables (Quinn and Keough 2002) . Raw data from the 6 trenches was tabulated to allow examination of patterns of water repellency and relative moisture content with soil depth. A Spearman's rank correlation test was undertaken on RWC and MED data to ascertain whether soil moisture was correlated with potential water repellency.

Experiment 3

Median MED values were calculated for depth of samples and distance from log and these were tabulated to allow examination of patterns. The rank based Kruskal-Wallis test was used to test whether water repellency varied significantly between depths and between distances from log. Multiple comparisons were made to determine which depths were different from each other using the "kruskalmc" test in the pgirmess package in R (Giraudoux 2010). Kruskal-Wallis tests were also undertaken on subsets of the depth data to test whether water repellency differed with

distance from log at individual depths. The frequency of water repellence (MED) categories in samples from eight ashbeds at five soil depths and at five distances from log were tabulated and graphed in order to assess the vertical and horizontal variability of water repellency.

A Spearman's rank correlation test was undertaken on distance to nearest seedling and surface layer (0-1cm) MED data to ascertain whether surface water repellency was correlated with distance to nearest seedling.

The relative water content data was square root transformed to improve normality. A 2 -way ANOVA was performed on the transformed data with the factors being depth and distance from log. A Spearman's rank correlation test was undertaken on RWC and MED data to ascertain whether was soil moisture content was correlated with potential water repellency.

4.3 Results

4.3.1 Experiment 1

There was three fold more water stored in surface soil (0-10cm) beneath a log than in adjacent bare ground. There was no significant difference in the relative water content of soil under burnt and unburnt logs. There was also no significant difference between water content of soil 5m away from logs in burnt and unburnt areas.

Table 4-1 Average relative water content of soil (± 1SE) under and 5m away from logs in burnt and unburnt areas of a eucalypt woodland at Liliesleaf Oatlands. Different letters represent significantly different means.

Treatment	Position	RWC (%) ± SE
Burnt logs (n=5)	under	11.5 ± 0.6 a
	5m away	3.5 ± 0.4 b
Unburnt logs (n=5)	under	12.5 ± 0.7 a
	5m away	4.6 ± 0.5 b

4.3.2 Experiment 2

Descriptions of the soil at different depths as sampled in log1 trench are shown in Table 4-2.

Table 4-2 Soil profile description from trench dug at log1 at Forton, Epping Forest

Sample Depth (cm)	Profile Description
0-2.5	A_0 horizon: ash,charcoal, burnt and unburnt litter over light grey light sandy loam
2.5-5	A_1 horizon: fine dark grey gravelly sandy loam
5-10	A_1horizon: grey gravelly sandy loam
10-20	A_2 horizon: light grey sandy gravel
20-40	B_1 horizon: dark orange/brown gravel
40-55	B_1 horizon: dark orange/brown gravel
55-80	B_2 light brown sandy clay to clay
>80	B_2 light brown clay

There was a strong correlation between MED class and WDPT class in dry samples ($r=0.921$, $p<0.001$). Therefore only MED results will be presented here. Water repellency MED tests (Table 4-3) showed that surface soils (0-2.5cm) in both unburnt and burnt areas without logs had severe to extreme water repellency while surface soils near burnt logs had no to low repellence, although both near log surface samples were quite variable at this coarse scale. All soil samples at depths from 2.5 – 5 cm showed severe to extreme repellency. Repellency graded from severe to low as depth increased and all samples were wettable by 10-40cm depth (Table 4-3). All samples taken below 30cm near and away from log1 were wettable.

Table 4-3 MED classes and water repellency categories of soil samples at different depths from trenches next to and 5m away from two logs in burnt areas and one log in an unburnt patch at Forton, Epping Forest

Depth (cm)	near log 1	away log 1	near log 2	away log 2	near log unburnt	away unburnt
0-2.5	1	10	4	10	9	10
2.5-5	10	10	10	9	8	9
5-7.5	8	8	9	7	8	7
7.5-10	8	7	7	6	7	6
10-15	7	6	4	4	3	1
15-20	7	5	2	2	1	1
20-30	4	4	1	1	1	1
30-40	1	1				

Soil water repellency categories

wettable	low	moderate	severe	extreme
1	2--4	5--7	8--9	10--11

There was no correlation between MED class and relative water content across all samples ($r = 0.239$, $p = 0.128$). Relative water content of surface soils soil near recently burnt logs was low compared to soil further away but water content of soil near an unburnt log was slightly higher than away from the log (Table 4-4). All upper areas of soil were dry compared to the clayey subsoil sampled below 70cm.

Table 4-4 Relative water content (%) of soil samples at different depths from trenches next to and 5m away from two logs in burnt areas and one log in an unburnt patch at Forton, Epping Forest

Depth (cm)	near log 1	away log 1	near log 2	away log 2	near log unburnt	away unburnt
0-2.5	0.44	1.01	0.61	1.61	3.59	1.94
2.5-5	0.53	1.63	0.94	2.53	3.55	1.87
5-7.5	0.25	1.88	1.46	1.61	1.44	2.44
7.5-10	0.24	1.60	1.02	3.22	1.16	1.77
10-15	0.31	1.19	2.29	3.53	0.80	1.19
15-20	0.99	0.96	1.75	2.24	0.69	1.01
20-30	1.47	0.72	0.67	0.91	0.51	3.33
30-40	0.65	0.54				
40-50	0.74	0.67				
50-60	1.66	1.51				
60-70	2.89	3.09				
70-80	8.06	9.19				

The finer scale measurements made in the field around a single partially burnt log revealed wettable soil in the top centimetre of soil up to 60 cm from the log (Table 4.5). At 30 cm from the log the soil was wettable down to 2cm. This wettable layer was underlain by soil of severe to extreme repellency down to 5cm depth where the soil became wettable again. Further away from the log at least the top 3cm of soil were severely to extremely repellent.

Table 4-5 MED class and water repellency categories at 5 soil depths and 6 distances from log. Measurements made *in situ* on one log only

	Distance from log					
Depth (cm)	0 cm	30cm	60cm	90cm	120cm	200cm
0	1	1	1	10	10	9
1	1	1	1	10	10	10
2	8	1	10	10	10	10
3	10	10	1	8	10	11
4	8	10	1	1	8	11
5	1	1	1	1	1	10

Soil water repellency categories

wettable	1	low	2--4	moderate	5--7	severe	8--9	extreme	10--11

4.3.3 Experiment 3

There was a significant difference in potential soil water repellency between soil depths ($\chi^2 = 27.84$, df 4, $p < 0.001$) but not at distance from log ($\chi^2 = 8.57$, df 5, $p=0.073$), when data from eight ashbeds was combined. The first and fifth centimetres below the surface were not significantly different in wettability and the second to fourth centimetres were significantly more repellent than the top centimetre (Figure 4-6). There were no significant differences in water repellency at different distances from log at each individual soil depth sampled.

The top centimetre of soil was wettable up to 80cm from the log and this wettable layer was underlain by a severely repellent layer of 2 to 3 cm in depth (Table 4-6). Adjacent to the log, this layer of severe repellency began at either 1-2 or 2-3cm below the surface. The soil generally became wettable again at 5cm depth close to the log. At 110cm from the log the pattern of repellency was distinctly different to that closer to the log with moderate repellence in the surface layers and low repellence below.

Table 4-6 Median MED class and water repellency categories at 5 soil depths and 5 distances from log. Samples were taken near logs in eight ashbeds. Soil depths that had significantly different median water repellency (MED) are indicated by different lower case letters

	Distance from log (cm)				
Depth(cm)	0	20	50	80	110
a 0—1	1	1	1	1	6.5
b 1—2	5	9	9.5	9.5	7
b 2—3	9	9	9	8	4
b 3—4	6	8	9	6.5	2
ab 4—5	1.5	5	7	5	1.5

Soil water repellency categories

wettable 1 low 2--4 moderate 5--7 severe 8--9 extreme 10--11

The variability of water repellence among the eight ashbeds sampled at the different depths and distances from log is shown in Figure 4-4. Water repellency was consistently absent in the top centimetre of soil next to the log in every ashbed sampled. Variability was greatest furthest away from the log with less than half of the eight surface layer samples being wettable 110cm from the log (Figure 4-4). An occasional sampling point from an individual ashbed was wettable down through each depth sampled, possibly representing a part of a vertical preferential pathway of wettable soil.

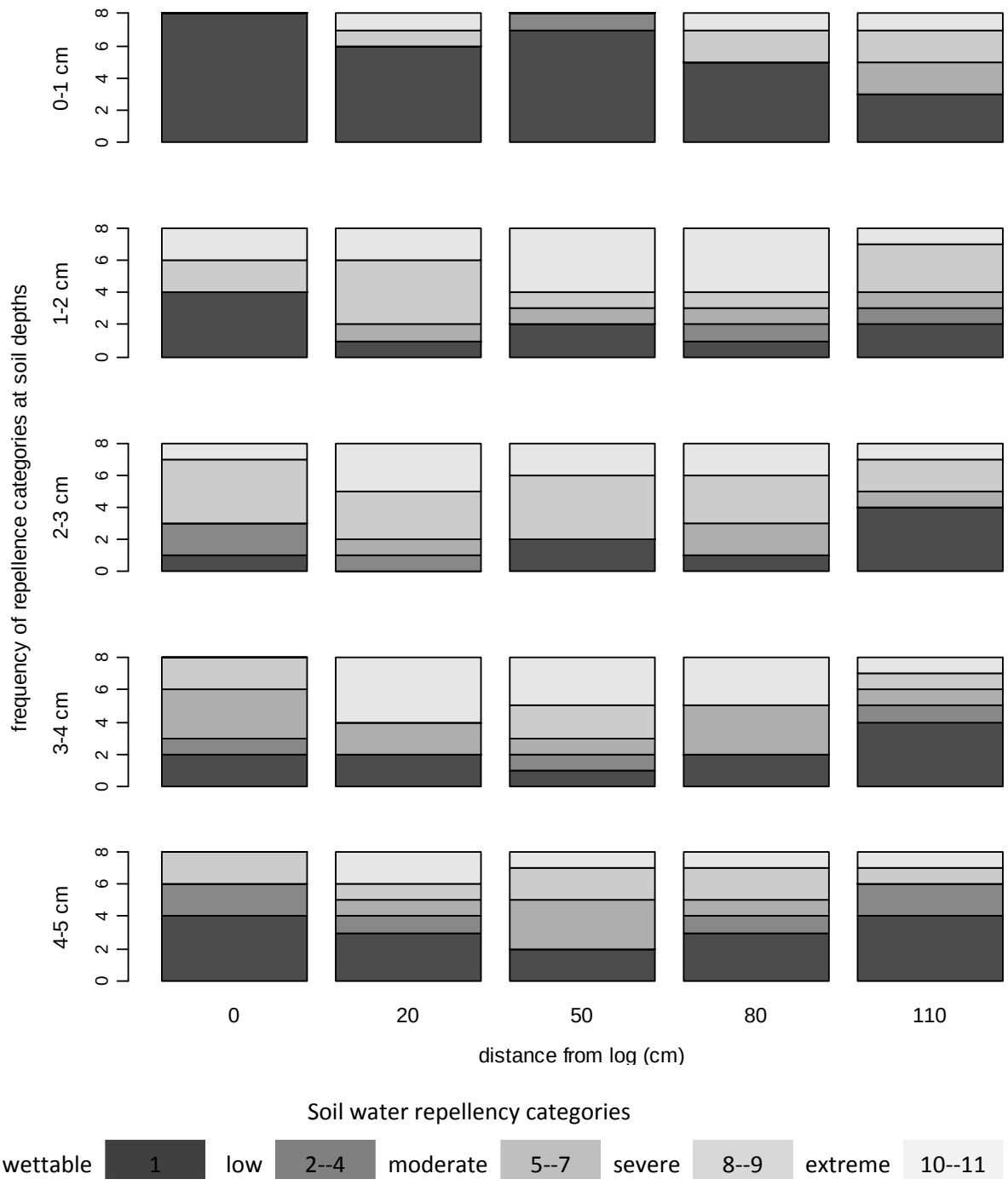

Figure 4-4 Frequency of soil water repellency categories in eight ashbeds at five soil depths and at five distances from log. Each box represents how many of the eight ashbeds sampled had soils in each water repellency category (shown by colour as per the legend) at a particular depth and distance from a log.

Horizontally, the surface layer of soil was the least variable with most of the samples taken up to 80cm away from each log being wettable (Figure 4-4). However there were some ashbeds in which surface samples were severely and extremely repellent

at 30cm, 80cm and particularly 110cm away from a log. This shows that around some logs the wettability of the surface of the ashbed was fairly continuous but was patchier around others. The most consistently repellent layer across all ashbeds was at 2-3cm (Figure 4-4).

Surface layer water repellency had a significant positive correlation with distance to nearest seedling ($r = 0.353$, $p= 0.013$). Soil water repellency (MED class) tended to increase with increasing distance from seedlings.

There was a significant difference in relative water content of the soil at different depths ($F_4=32.1$, $p<0.001$) and at different points ($F_4 =3.1610$, $p= 0.015$). At the time of sampling the top centimetre of soil was significantly wetter than the 1-2cm layer which had significantly higher water content than the two lowest layers but not the 2-3cm layer (Figure 4-5).

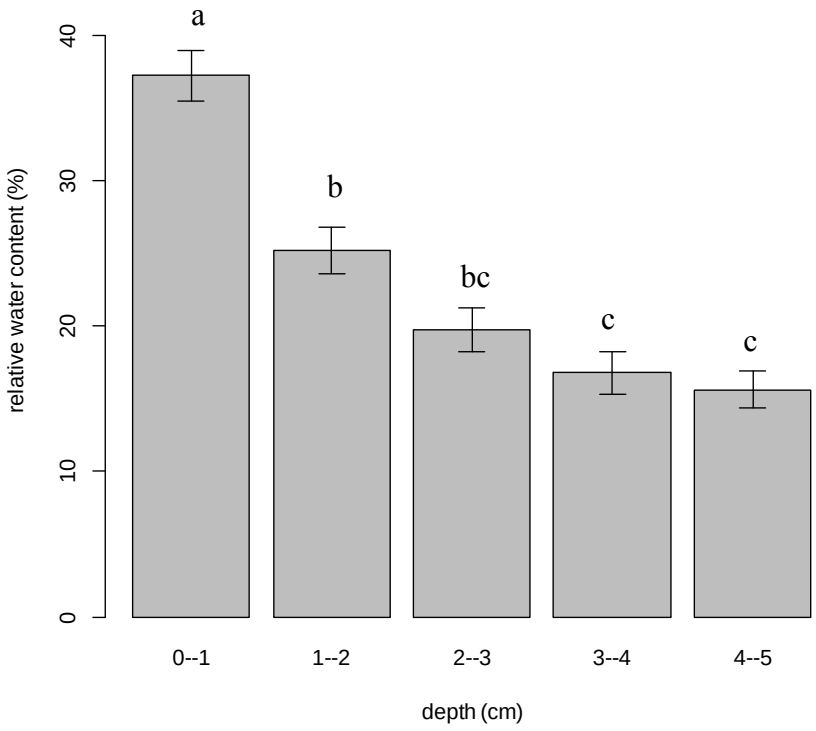

Figure 4-5 Mean relative water content (+/- 1SE) of soil at 5 different depths across eight ashbeds. Different letters represent significantly different means.

Across all ashbeds the soil 20 cm from the logs (average 20.27% RWC) was significantly drier than the soil 110cm from the log (average 26.83% RWC). All other distances were not significantly different from each other. There was a significant negative correlation between RWC and soil water repellency (MED class) ($r= -0.4073$, $p<0.001$). As the potential soil water repellency increased the moisture content of the soil tended to decrease.

4.4 Discussion

The soil water repellency characteristics of typical eucalypt seedling microsites (in ashbeds near coarse woody debris) showed the classic pattern of soil heated under high intensity fire as described by DeBano (2000), Letey (2001), Doerr *et al.* (2004) and Shakesby *et al.* (2007). Next to partially burnt logs, soil was wettable for 1-2 centimetres below the surface and this wettable layer was underlain by a severely water repellent layer of at least one centimetre thickness with less repellent soil below that. At half of the logs sampled, the soil was wettable again by 5cm depth. In areas of less intense and no burn (outside ashbeds and in unburnt soil) the surface layers were moderately to extremely water repellent, as would be expected in soil from a dry eucalypt forest (Crockford *et al.* 1991; Doerr *et al.* 2006; Howell *et al.* 2006).

The horizontal pattern of water repellency across ashbeds was fairly consistent in this study. Therefore, the analysis of median water repellency (MED classes) showed no difference in distance from log. A similar horizontal layered pattern of wettable soils over severely to extremely repellent soils over less repellent soil was common across the ashbeds studied, although there was some patchiness. This is similar to post fire

water repellency patterns reported by DeBano (2000), Doerr *et al.* 2006 and Shakesby *et al.* (2007).

There was however, greater variability in water repellency in soil measured furthest from the logs in comparison to that near logs. Far from logs not all surface samples were wettable or middle layers highly repellent. At the much greater distance from logs (5m) measured in Experiment 2, where samples were taken outside ashbeds, the surface soils showed severe repellency down to at least 5cm. This pattern of severe surface soil repellency was described by Shakesby *et al.* (2007) as typical of lightly burnt or unburnt soils in dry eucalypt forests.

In agreement with North American studies reported by Harmon *et al* (1986) and Amaranthus *et al.* (1989), logs were shown in this study to provide a reservoir of moisture in the soil directly underneath them. Soil had three fold greater water content compared to soil 5 metres away. This was the case whether the logs had been burnt or not. Measurements in Experiment 1 were taken less than a year after fire and in a relatively dry year and yet enough moisture had built up under the burnt logs to be similar to that under unburnt logs.

The moisture storing capacity of logs and the patterns of soil water repellency near and away from logs influences the availability of soil moisture and therefore has implications for seedling recruitment and survival. Seedling mortality is primarily a function of moisture stress in dry periods in dry forests and woodlands (Stoneman 1994; Yates *et al.* 1996; Ashton 2000; Khurana and Singh 2002). Fine scale trends in moisture availability can be important determinants of seedling survival and distribution patterns (Hobbs and Cramer 2003). Adequate surface moisture for seeds

to germinate as well as connectivity between wettable patches and soil water storage areas are required for seedlings to establish.

In Experiment 3, undertaken in winter after good rains, the wettable surface layer in ashbeds was significantly wetter than the 2cm deep potentially extremely repellent layer below it. Both these layers were wetter than the less repellent layers further below. Thus the wettable surface soil layer underlain by a relatively impermeable layer trapped moisture near the soil surface providing a moist seed bed for eucalypt seed to germinate. Germination was prolific where these conditions prevailed (in ashbeds). However, the fact that the soil from between 1-4cm depths that had average moisture contents of 15-25% when collected became severely to extremely repellent on air drying, has implications for the survival of the germinated seedlings as the soil begins to dry out over summer.

The highly repellent layer just a centimetre or so under the soil surface could disconnect germinated seedlings from moisture stored further down the profile, as described by Madsen (2010). In the current study, the soil 5cm down (despite being mainly wettable), had a significantly lower water content than the layers above it. This result shows that even in wet conditions, the water repellent layer can reduce the amount of water that infiltrates into the layers immediately below it. The repellent layer may cause excess water in the wettable layer above it to flow away horizontally as distribution flow (Dekker and Ritsema 1995), reducing the amount of water available at depth for the development of germinated seedlings.

Alternatively, the water repellent layer could allow the wettable layer above it to act as a sump for water that runs off any adjacent patches of soil with severe surface water repellency. Differences in water repellency horizontally along the soil surface

as well as microtopography are important here. In a study of the effects of slope and water repellency of substrates, Osborn *et al.* (1967) found germination of ryegrass seed was completely prevented in a non-wettable soil substrate when it was held on a slope. Seeds on a sloping substrate that was treated with wetting agent germinated at near the viability rate. Some but not all seeds were washed off the sloping treatments. Germination of seeds in the water repellent soil held level was delayed by four days relative to the seeds in the non sloping wetting agent treatment. Germination occurred on the water repellent substrate due to a ponding of water above the soil surface that slowly reduced the hydrophobicity and provided enough water for the seeds to imbibe.

The ponding of water in surface wettable patches over time can convert the repellent layer beneath to wettable and eventually induce a wettable finger of soil through which water can move (Doerr *et al.* 2000). This could occur at the edge of ashbeds at the interface between more and less intense fire. Indeed one sample point at the edge of an ashbed, 110cm from a log, was entirely wettable in the top 5 centimetres and a seedling was only 5cm away. Depressions in the soil surface could also be areas in which surface water ponds and wettable fingers develop (Dekker *et al.* 2009) however this was not specifically studied here.

In this study centimetre scale variations in water repellency occurred and, as shown by Dekker and Ritsema (2000), this can lead to the formation of preferential pathways and unstable wetting fronts. The proximity of a newly germinated seedling to a preferential pathway such as a finger of wettable soil or macropore may have a strong influence on its survival. Soils under and around eucalypt trees have been shown to be highly macroporous (Eldridge and Freudenberger 2005) and these macropores represent possible pathways for water to move through water repellent

soils. Macropores are biological hotspots in soil with better oxygen and nutrient status than bulk soil and studies have shown that living roots tend to grow preferentially in macropores (Stewart *et al.* 1999; Jarvis 2007).

Proximity of seedlings to preferential pathways or macropores was not determined in this study due to the complex nature of the underground system and the inherent difficulties of studying it. However, it is logical to conclude that seedlings that germinate near to or on top of macropores and wettable fingers of soil through which water flows to the subsoil, would have ready access to moisture. This would give these seedlings a greater chance of surviving periods of little rain than those seedlings surrounded by repellent soil which disconnects them from soil moisture. The temporal variability of water repellency is also important here and deserves further research.

The water repellency of soil around seedlings during dry periods is likely to act as an environmental sieve (*sensu* Harper 1977) and reduce the breadth of the seedling establishment niche compared to the germination niche. This study has shown that outside ashbeds the surface soil commonly has potential water repellency that is severe to extreme, but may have transient phases after extended periods of rain where soil is wettable and holds considerable amounts of water (up to 27% RWC). Seeds may germinate in these patches during wet periods but become cut off from soil moisture sources over time as the soil around them dries and becomes repellent.

Seedlings in ashbeds have the advantage of being in an extensive surface layer of wettable soil that remains wettable even in dry weather and provides storage for any precipitation. Thus wettable ashbeds around coarse woody debris constitute suitable patches on the forest floor that enable more individuals to make the critical transition

from germination through to establishment compared to patches with severe inherent surface hydrophobicity.

In exposed ashbeds however, particularly on sloping ground, an extensive area of wettable soil over a repellent layer may have detrimental effects. If rain water flows over the repellent layer and out of the ashbed during high intensity rainfall events, it may erode the wettable layer. This has been observed throughout the world in studies of post fire erosion on steep hill slopes and has been shown to be worst during heavy rain after extended dry periods (DeBano 2000; Ferreira *et al.* 2005; Shakesby and Doerr 2006; Fox *et al.* 2007; Nyman *et al.* 2010).

There have been reports of eucalypt seedlings having difficulty establishing in exposed ashbeds (McCormick 1991; ForestryCommission 1993; Florentine *et al.* 2008) and clusters of dead seedlings on sloping exposed ashbeds were observed at Tom Gibson Reserve, Epping Forest during the seedling survey described in Chapter 3 of this thesis. The interaction of an extensive subsurface water repellent soil layer with topography and position in the landscape may therefore influence seedling survival and distribution patterns. Similarly sloping areas with highly repellent surface soils may act as an environmental barrier that prevents seedling establishment all together (Osborn *et al.* 1967).

This highlights a possible role of coarse woody debris in the seedling recruitment niche. Coarse woody debris may act as obstructions to overland flow (Ludwig and Tongway 1996) and protect seedlings from the erosive effects of distribution flow over a water repellent soil layer. Seedlings that germinate in ashbeds near coarse woody debris may benefit from the associated high moisture storage capacity under a log and have the advantage of being able to connect to soil moisture throughout the

year. Being close to coarse woody debris has the added benefits for seedlings of protection from wind, temperature fluctuations and browsing animals as described in Chapter 3 of this thesis.

The current study specifically targeted logs at the centre of ashbeds, however smaller pieces of woody debris could also influence soil water repellency. A number of sampling points near partially burnt branches laying on a larger ashbed also showed wettable surface soils. It is therefore possible for much smaller ashbeds to also provide ideal conditions for seedling germination and establishment and to add to the heterogeneity of water repellency across the soil surface. A finer scale of measurement than that undertaken in this study may better elucidate the heterogeneity of water repellency across fire intensity boundaries over the forest floor.

A finer scale of sampling may also further illuminate the patterns of water repellency and soil moisture near and moving away from seedlings. In this study, surface soil closer to seedlings had a significant tendency to be less repellent than that further away. More targeted measurements near and moving away from seedlings would more clearly describe the water repellency conditions experienced by seedlings and moisture sources for them wherever they may be on the forest floor.

Despite the short comings inherent in the scale of this study, it is clear that soil water repellency in dry Tasmanian forests is highly variable both vertically and horizontally in accordance with the findings of Howell *et al.* (2006) and that this is affected by fire intensity as measured by the proximity to burnt heavy fuels (DeBano 2000; Doerr *et al.* 2000; Letey 2001). Both the spatial and temporal heterogeneity of

water repellency has important implications for eucalypt recruitment and persistence that deserve further exploration.

4.5 Conclusion

Inherent water repellency in eucalypt woodland soils promoted by hydrophobic substances from adult trees could play a large role in the restriction of eucalypt recruitment in inter fire periods. The interaction between fire and coarse woody debris and the resultant intense soil heating and formation of ashbeds, leads to spatially restricted patches on the forest floor in which surface water repellency is reduced, moisture accumulates and seedling germination and establishment is promoted.

The spatial and temporal variability of soil water repellency described in this study and its possible effects on the three dimensional movement and storage of water in the soil, could act as environmental sieves that selectively filter out some individual seedlings while allowing others to pass through to the persistence phase. The elevated amounts of soil moisture under logs reported in this study represents a significant reservoir of water for seedlings that have germinated nearby and could be a major attribute of a germination microsite that allows a seedling to become established, survive dry periods and continue with its ontogeny. Similarly, macropores and fingers of wettable soil could supply a relatively constant moisture supply to seedlings established nearby and allow them to persist through dry periods that cut off and kill seedlings further away from the moisture source. Future studies undertaken at finer scales than this one may further elucidate these processes.

The findings of this study were made after the establishment of the restoration treatments described in the following two chapters. Thus the understanding of the

importance in the eucalypt recruitment niche of fire effects on, and heterogeneity of, water repellency was not built into the restoration trial experimental design. Indeed it was only upon the watering of seed and planted seedlings in treated plots that observations of the differences in water infiltration between burnt and cultivated plots were made and the implications of water repellency first realised. However insights gained from this finer scale study can be used to interpret results of the restoration trials.

Chapter 5 Restoring the eucalypt recruitment niche

5.1 Introduction

Many woodland remnants in the agricultural districts of Tasmania are not viable in the long term due to lack of regeneration, as discussed in Chapter 2 of this thesis. These remnants represent relict populations temporarily existing as non-recruiting adults (Eriksson 2000; Young *et al.* 2005). There is a need to restore eucalypt regeneration in these remnants to optimise the viability of remaining treed areas (Close and Davidson 2003) or otherwise face the possibility of a treeless landscape by the end of the century (Gibbons *et al.* 2008b; Fischer *et al.* 2009). However, crucial elements of the recruitment niche (as described in Chapter 3) may be missing in these remnants and thus act as barriers to the establishment of new seedlings.

The degrading processes that reduce the structural complexity and health of remnants, and restrict regeneration in them, have important implications for the success of restoration attempts using direct sown seed and/or planted seedlings (Close and Davidson 2003). The regeneration dynamics of degraded sites almost certainly differ from those of less modified sites (Suding *et al.* 2004). Additionally, the transition of degraded ecosystems with little or no regeneration back to more desirable states (i.e. with population replacing quantities of regeneration), may be prevented by restoration barriers (biotic, abiotic or both) that need addressing before attempting revegetation (Hobbs and Harris 2001, Chapter 1). This chapter explores potential restoration barriers and the measures needed to overcome them in order to restore the eucalypt recruitment niche in degraded woodland remnants.

Suding *et al.* (2004) suggest once restoration goals have been set, the constraints that may represent restoration barriers in the degraded system need to be identified then prioritised. Ideally the efficacy of remedial actions on these constraints needs to be tested, as conducted in this study and reported in this and the following chapter.

5.1.1 Potential barriers to recruitment in degraded remnants

Possible constraints in degraded systems include disturbance regime; physical (abiotic) conditions; changed above and below ground interactions; herbivory; propagule limitation; and regional environmental change (Tommerup and Bougher 2000; Suding *et al.* 2004).

Disturbance regime: In agreement with other studies in south east Australia and in the south west of Western Australia, the work described in Chapter 2 found that the intensity of livestock grazing was one of the most important degrading influences on eucalypt regeneration in Tasmanian remnant woodlands and that lack of fire was another important barrier to recruitment. It is generally accepted that the frequency of fire has decreased, and the intensity and scale of fire has increased since European settlement in temperate areas (Bowman 1998; Close *et al.* 2009a). A shift in mean fire frequency can induce shifts in vegetation composition and structure and soil physical and nutritional characteristics and this has consequences for adult tree health, fuel accumulation and eucalypt recruitment (Jackson 1968; Close *et al.* 2009a).

Tasmanian households are estimated to use over half a million tons of firewood annually (RPDC 2006a). Firewood collection is undertaken almost exclusively in dry sclerophyll forests and woodland remnants in Tasmania, often on private land and with *Eucalyptus amygdalina* extensively targeted (Miller 2001; RPDC 2006a). This

represents a major degrading influence in these remnants and removes heavy fuels which would otherwise contribute to hot spot fires.

Abiotic conditions: Nutrient enriched and compacted soils are legacies of past applications of fertilizers and heavy grazing (Yates *et al.* 2000b; Prober *et al.* 2002; Close *et al.* 2008; Duncan *et al.* 2008). A study by Close *et al.* (2008) in remnants in the current study area showed that recently heavily grazed remnants had raised levels of Nitrate N and Colwell P and significantly less penetrable soils than in healthier remnants and that this impacted on adult tree health. The C/N ratios in the heavily grazed remnants were in the range of typical agricultural soils managed with fertilisers while those in lightly grazed remnants were typical of soils of healthy temperate eucalypt woodlands and forests (Granger *et al.* 1994). Fertiliser drift from adjacent agricultural land also increases nutrient enrichment, particularly in small to medium sized remnants and around the edges of large remnants and this could impact seedling regeneration (Duncan *et al.* 2008). However, one of the remnants surveyed in the recruitment niche study was nutrient enriched with significantly higher N, P, K and S than the three other nature reserve sites studied and yet it contained plentiful regeneration in the form of seedlings and lignotuberous sprouts (Barton Farm site, reported in Chapter 3). This suggests that nutrient enrichment alone may not be a barrier to regeneration but it is implicated in its interactions with biotic components of degraded ecosystems (Close *et al.* 2008; Skinner *et al.* 2010).

Compaction of soils by livestock and machinery leads to decreased penetrability, increased bulk density, reduced water infiltration and increased runoff (King and Hobbs 2006), all of which has implications for water availability and root penetration for germinating seeds and growing seedlings (Passioura 1991; Yates *et al.* 2000b; Skinner *et al.* 2009). Skinner *et al.* (2009) showed that eucalypt seedlings were more

susceptible to surface drying in compacted soils and thus at greater risk of desiccation following germination. Lack of surface roughness in both compacted and uncompacted soils may also affect water availability and suitability as a seed bed (Battaglia and Reid 1993; Yates *et al.* 2000b). A disturbed seed bed is required as eucalypt seed is small, there is limited or no endosperm and the radicle needs to find a way into moist soil rapidly in order to take up water and nutrients (Florence 1996).

Soil water repellency is another potential barrier to eucalypt recruitment, both in healthy and degraded sites, as discussed in the previous chapter. Spatial and temporal heterogeneity in soil water repellency influences where and when seeds are able to germinate and seedlings establish. Unburnt soils in eucalypt woodlands and forests can be severely hydrophobic (Chapter 4, Doerr *et al.* 2004). Subsequent preferential flow of water as well as lack of wettable patches (initiated by intense fire in coarse woody debris) may restrict eucalypt recruitment.

Coarse woody debris was also shown to be a vital element of the seedling recruitment niche with functions in the provision of shelter, microclimate amelioration, moisture sinks and protection from browsing (Chapter 3). Intermediate sites measured in the study reported in Chapter 2 had, on average, two thirds the length of logs ≥10cm in diameter found in healthy sites (Table 2-9). Lack of nurse structures such as logs and branches that facilitate a favourable microsite condition for establishment of seedlings may represent another barrier to recruitment.

Above and below ground interactions: Cramer *et al.* (2006) suggest that the legacy of agricultural practices shifts establishment niche availability to favour the development of novel plant communities comprised of species adapted to the altered environmental conditions. Often these novel communities are made of a "recalcitrant

understorey layer" that includes invasive native species and/ or unwanted exotics (Royo and Carson 2006). In the Midlands *Lomandra longifolia* and *Pteridium esculentum* (bracken fern) are two native species that have traits that have enabled them to adapt and thrive in nutrient enriched agricultural environments (Cramer *et al.* 2006; Mokany *et al.* 2006). They have become regarded as pest species by farmers as they are relatively unpalatable to livestock, can blanket the understorey of remnant tree stands and invade pasture (Mokany *et al.* 2006; McWhirter and Kemp 2010). These species are local examples of what Young *et al.* (2010) call "widespread and pervasive modifiers of ecosystems and disruptors of forest regeneration" that may also act as ecological filters allowing some species to regenerate and not others (George and Bazzaz 1999). The widespread presence of these species in the understorey of degraded remnants (as was the case in many of the remnants of intermediate condition in the study described in Chapter 2) could constitute a biotic barrier to eucalypt recruitment.

Other persistent species in the understorey in degraded remnants in the Midlands and elsewhere that may compete for space and resources with eucalypt seedlings include exotic grass, herbaceous and woody weeds. Nutrient enrichment in remnants can favour the establishment, competitive ability and persistence of these exotic species with the possibility of positive feedbacks maintaining their populations in preference to native species (Prober *et al.* 2002; Suding *et al.* 2004; Cramer *et al.* 2006; Standish *et al.* 2008).

Woodland fungi are very susceptible to disturbance (Tommerup and Bougher 2000). Ectomycorrhizal fungi associated with roots of eucalypts are known to improve early growth by increasing the uptake of water and of phosphorus (P), especially where P is limiting (Bougher *et al.* 1990). The formation of ectomycorrhizae in eucalypts and

their positive effects on nutrient uptake and plant growth are strongly inhibited by excess amounts of available N and P (Mason *et al.* 2000). Other adverse disturbance effects such as loss or alteration of top soil, litter and organic matter and a reduced complement of host plant species decreases the diversity of mycorrhizal fungal communities and the inoculum levels of ectomycorrhizal fungi (Tommerup and Bougher 2000). The growth of eucalypts is inhibited in soils without ectomycorrhizal inoculum (Ellis and Pennington 1992; Close and Davidson 2004) and lack of ectomycorrhizal fungi in degraded woodlands may therefore play a role in limiting eucalypt establishment.

Herbivory: The Tasmanian Midlands have large populations of native brushtail possums (*Trichosurus velpecula*) with their numbers having steadily increased since the collapse of the fur trade in the 1980's (Neyland 1996). Possums are generalist herbivores that have been implicated in the decline in adult eucalypt tree health (Neyland 1996; Kirkpatrick *et al.* 2000) and are one of a number of marsupial browsers that cause significant damage to eucalypt seedlings in forestry operations (Miller *et al.* 2008). Large populations of mammal herbivores (including feral animals such as deer, hares and rabbits) could constitute a significant barrier to the persistence of eucalypt seedlings in degraded woodland remnants. Insect herbivory has also been implicated in the decline in health of mature trees in nutrient enriched remnants elsewhere (Landsberg *et al.* 1990) and may also constrain the survival and growth of seedlings in the Midlands.

Propagule limitation: Eucalypts are weakly serotinous with a continuous, low level of seed release occurring throughout the year that is accelerated by hot dry conditions or fire. This leads to a temporally variable seed rain. Dispersal of eucalypt seed is limited with passive release of seed aided by wind (House 1997). Predation by ants

usually prevents the formation of a soil seed bank (Yates *et al.* 1995) and may substantially reduce seed available for germination (Clarke 2000; Clarke and Davidson 2001). Seed availability is therefore spatially and temporally variable (Yates *et al.* 1994a) and may constrain natural eucalypt regeneration (Clarke and Davidson 2001).

Regional environmental change: Meteorological data indicate that average annual rainfall over the past 30 years in the Midlands of Tasmania (Oatlands 42°18'S, 147°22'E) has been 498 mm compared to the long-term average of 551 mm. There has been a change in year to year variability in rainfall since 1975 and the seasonal pattern of rainfall has also changed, from a more even distribution to a pronounced dry period in late summer/autumn (Kirkpatrick *et al.* 2000; ACECRC 2010). Tasmanian temperatures have risen since the 1950s, but at a slower rate than in mainland Australia (ACECRC 2010). These changes have serious consequences for moisture availability for seedlings over summer and suggest that seasonal moisture deficit may be an important limiting factor in eucalypt recruitment.

5.1.2 Potential methods of addressing constraints on regeneration

Yates *et al.* (1994) described the role of large scale natural disturbances such as fire, wind storms, floods and drought in creating recruitment sites for woodland eucalypts. This was reinforced by the results of the microsite survey in Chapter 3 of this thesis which showed the importance of fire and resultant ash beds in the recruitment niche of woodland eucalypts in Tasmania. However, the use of fire as a broad scale restoration treatment has implications for other aspects of conservation management as it may have undesirable impacts on remnant trees, any existing regeneration,

habitat provision and spread of exotic species (Lockett and Candy 1984; McCormick 1991; Gilfedder and Kirkpatrick 1998; Yates *et al.* 2000a). Similarly, techniques developed for large scale reforestation in plantations for timber and environmental plantings on farms (Close and Davidson 2003; Davidson and Close 2006), while offering ideas for restoration methodology, may also not be wholly appropriate to the scale of a remnant stand. For example, major soil disturbance, broad scale herbicide and fertiliser use may have ramifications for remnant trees, other native species and habitat structures. Yates *et al.* (2000) declared that "the challenge for restoration ecologists is to see if large scale disturbances such as fire can be turned into small scale management activities which do not endanger a whole remnant population". The current study aims to do this by trialling patch scale treatments within existing remnants.

Numerous other studies have shown that establishing eucalypt regeneration in grassy understorey and grazed land is extremely difficult (Ellis and Pennington 1992; Fensham and Kirkpatrick 1992; Pinkard 1992; Clarke and Davidson 2001; Clarke 2002; Li *et al.* 2003; Semple and Koen 2003; Skinner *et al.* 2009). Natural recruitment usually only occurs when a weed-free seedbed has been deliberately or accidentally prepared through some sort of disturbance followed by sufficient rainfall (Venning 1988; Curtis and Wright 1993; Cluff and Semple 1994; Semple and Koen 2003). The study in Chapter 3 showed microsite attributes that promote soil moisture are critical.

This study addresses some of the above constraints to recruitment articulated above through the experimental application of the following:

1. **Patch scale plots in canopy gaps:** As shown in Chapter 2, healthy remnant vegetation is structurally complex and heterogeneous whereas degraded remnants have been simplified in structure and composition and therefore tend to lack the patchiness of resources displayed in healthy ecosystems (Dorrough *et al.* 2006). Patch scale restoration treatments may improve heterogeneity within a stand (Tommerup and Bougher 2000) and localize water movement (Hobbs and Cramer 2003; King and Hobbs 2006) while reducing potential damage to existing trees and regeneration. Canopy gaps (where resource competition from adult trees is likely to be least (Battaglia and Wilson 1990)), were previously identified as an essential element in the recruitment niche (Chapter 3).
2. **Fire:** Creating an intense burn at a patch scale through piling up and burning coarse woody debris (similar to forestry debris piles or "turkey heaps" that are burnt to create ash beds) may have positive effects towards reducing a number of the constraints caused by a lack of fire. These include lack of seed bed, soil water repellency and competition from recalcitrant understorey species (native and exotic). The high surface soil temperatures reached during intense fire have also been shown to change microbial populations and processes in the soil such that seedling growth is stimulated (Renbuss *et al.* 1973; Chambers and Attiwill 1994; Florence 1996)
3. **Cultivation and herbicide:** Soil improvement through careful cultivation may eliminate or ameliorate problems associated with compaction, lack of surface roughness (Tongway and Ludwig 1994) and poor seedbed conditions while the concurrent use of herbicide to kill existing weeds may address biotic

competition constraints (Pinkard 1992; Yates and Hobbs 1997a; Yates *et al.* 2000a; Close and Davidson 2003; Davidson and Close 2006).

4. **Coarse woody debris:** Addition of large logs (which are an important element of a healthy eucalypt seedling microsite) may overcome the problems associated with loss of water resources, lack of nurse structures and lack of fuel for intense spot fires.
5. **Seed addition:** Addition of seed collected from local healthy trees bypasses the reproduction and dispersal niches (Young *et al.* 2005) and alleviates problems of lack of viable propagules. Plots were also open to local seed rain.
6. **Initial irrigation:** the application of cold stratified, pre-primed eucalypt seed (soaked in water prior to sowing) and initial watering of applied seeds *in situ* may overcome initial constraints on germination caused by innate dormancy and/or lack of reliable rainfall (Battaglia 1996; Close and Wilson 2002).
7. **Fencing:** Fences were placed around all individual plots and most of the remnants (where necessary) in an attempt to address constraints caused by livestock grazing and native and feral animal browsing.

This study aims to overcome some of the previously identified barriers to eucalypt regeneration by using techniques designed to test and recreate the eucalypt recruitment niche.

The research questions of this study were identified as follows:

1. Does site amelioration through added fire or cultivation and herbicide treatment overcome apparent barriers to eucalypt recruitment in these sites?
2. Does using or enhancing the reported elements of the natural eucalypt recruitment niche (fire, ashbeds, coarse woody debris and local seed) improve

eucalypt germination and establishment compared to seedbed preparation through cultivation?

3. Does the condition of a site, as measured by structural complexity, influence the amount of eucalypt recruitment after seed bed preparation through fire or cultivation?
4. Do these treatments produce sufficient regeneration for population replacement in degraded remnants?

5.2 Methods

5.2.1 Experimental sites

Two experimental sites were located at each of three locations within the Midlands of Tasmania, as follows:

1. **Epping Forest in the Northern Midlands** Tom Gibson Reserve and the adjacent Downie sheep/ cropping property "Valleyfield".
2. **Fingal in the eastern Northern Midlands** Gunns' Evercreech coupe and an adjacent remnant surrounded by plantation that was previously grazing land.
3. **Oatlands in the Southern Midlands** Meaburn sheep/cattle property "Liliesleaf" and the area of the adjacent Weeding sheep property "Weedington" known as "Bald Hill".

Descriptions of the sites are given in Table 5-1and mean and standard error values for structural complexity attributes (measured for the study described in Chapter 2 of this thesis) are given in Table 5-2. A lightly grazed healthy remnant and an adjacent heavily grazed degraded remnant were chosen at Epping Forest and Fingal in order to test whether the effects of establishment treatments on germination and seedling survival were influenced by the initial health of the experimental site. At Oatlands the two sites were both in a similarly degraded state with the Liliesleaf remnant having a slightly higher structural complexity score than Bald Hill (56.8 vs 53.5 Table 5-2). These sites were also a part of a larger study across a range of forest types in the Southern Midlands (funded by NRM South) that focused on creating demonstration sites of restoration treatments to arrest rural tree decline in degraded native vegetation remnants and no healthy remnants were included.

Table 5-1 Description of the six research sites including remnant *a priori* health class (Chapter 2), grazing history, TASVEG vegetation community, date plots were treated (burnt or cultivated) and date plots were sown with eucalypt seed and planted with eucalypt seedlings.

Site (abbreviation in Chapter 2)	Latitude Longitude	Class	Grazing history	TASVEG comm.#	Comments	Treatment date	Sowing date
Epping Forest – Northern Midlands							
Tom Gibson Reserve West (WTG)	41.779°S 147.290°E	Healthy	Light	DAZ	Long unburnt section of reserve with native grass, low shrub and *Acacia* understorey	14 May 2008	19 Sept 2008
Valleyfield remnant (DD)	41.787°S 147.311°E	Intermediate	Heavy	DAZ	Surrounded by improved pasture, fenced off in 2000, with pasture and native grasses, *Lomandra*, bracken, low native shrub and *Acacia* understorey. Some patchy gravel extraction.	14 May 2008	24 Sept 2008
Fingal – Eastern Midlands							
Evercreech coupe (EC)	41.510°S 147.956°E	Healthy	Light	DAM	Recently partially harvested coupe with residue heaps immediately adjacent to healthy ungrazed remnant with native grass herb and shrub understorey	20 May 2008	10 Oct 2008
Evercreech plantation remnant (ED)	41.529°S 147.981°E	Intermediate	Heavy	DAM	Remnant surrounded by 5 year old plantation with pasture grass, bracken, exotic weed understorey, crash grazed	20 May 2008	29 Sept 2008
Oatlands – Southern Midlands							
Liliesleaf remnant (M)	42.319°S 147.406°E	Intermediate	Heavy	DPO	Farm remnant fenced in 2007 for NRM south project with native and pasture grass, *Lomandra*, bracken, weedy understorey	26 May 2008	16 Oct 2008
Bald Hill remnant (BH)	42.309°S 147.407°E	Intermediate	Heavy	DPO	Farm remnant fenced in 2007 for NRM south project with native and pasture grass, *Lomandra*, bracken, weedy understorey	26 May 2008	15 Oct 2008

*Average annual rainfall is based on long term records (starting 1882at Fingal and Oatlands and 1923 at Epping Forest) at the closest weather station to the sites with the number in brackets being the Bureau of Meteorology station number (http://www.bom.gov.au/climate/data/). # TASVEG vegetation community type: DAZ - *Eucalyptus amygdalina* inland forest and woodland on Cainozoic alluvial deposits, DAM - *Eucalyptus amygdalina* forest and woodland on mudstone , DPO - *Eucalyptus pauciflora* forest and woodland not on dolerite substrates (http://www.dpiw.tas.gov.au/inter.nsf/webpages/ljem-6rf6jf?open) Soils at Oatlands are sandy loams over Triassic sandstone bedrock. Soils at Fingal are on Mathinna beds which weather to sandy soils of low fertility(Laffan *et al.* 1998).

Table 5-2 Means and standard errors (in brackets) of structural attributes measured in three 50 x20m quadrats at each site; ratio of regenerating stems to adult trees; and structural complexity index scores (see Chapter 2) for the six research sites.

Site	Tom Gibson Reserve West	Downie Valleyfield Farm Remnant	Evercreech Coupe	Evercreech Plantation Remnant	Weedings Bald Hill Farm Remnant	Meaburns Liliesleaf Farm Remnant
Location	Epping Forest	Epping Forest	Fingal	Fingal	Oatlands	Oatlands
Basal area m^2ha^{-1}	24.7 (2.7)	12.8 (3.2)	41.1 (3.9)	26.3 (3.9)	20.6 (6.1)	19.6 (10.2)
Quadratic mean dbh cm	44.5 (5.8)	42.0 (12.6)	60.2 (2.3)	90.9 (7.1)	67.1 (18.4)	65.8 (14.3)
No. trees>40cm ha^{-1}	63.0 (18.6)	16.7 (8.8)	123.3 (8.8)	36.7 (3.3)	43.0 (18.9)	36.7 (17.6)
No. dead trees ha^{-1}	80.0 (23.1)	13.3 (3.3)	13.3 (3.3)	16.7 (6.7)	20.0 (3.5)	26.7 (8.8)
No. hollow bearing trees ha^{-1}	23.3 (6.7)	13.3 (6.7)	13.3 (3.3)	20.0 (0)	3.3 (2.2)	16.7 (6.7)
No. perennial species	17.3 (1.5)	15.0 (0.6)	23.0 (1.5)	9.3 (0.9)	12.3 (3.8)	11.7 (0.3)
No. life forms	7.3 (0.3)	8.0 (0.6)	8.3 (0.3)	6.0 (0)	7.0 (2.3)	7.0 (0)
Vegetation cover <0.5m	75.0 (4.4)	69.6 (4.6)	71.3 (14.2)	77.1 (7.1)	48.8 (16.4)	75.9 (2.0)
Vegetation cover 0.5- 6m	11.8 (4.1)	28.1 (14.0)	16.8 (10.4)	2.5 (2.5)	5.6 (2.7)	0.0
Total length of logs mha^{-1}	2043.3 (276.3)	1503.0 (363.4)	1555.0 (190.2)	1301.7 (325.7)	855.0 (244.2)	626.7 (299.0)
Length of large logs mha^{-1}	203.0 (63.6)	345.0 (54.8)	391.7 (151.9)	286.7 (101.8)	110.0 (80.6)	188.3 (98.0)
Dry weight of litter tha^{-1}	10.8 (1.3)	10.3 (2.1)	14.4 (1.5)	8.5 (1.0)	9.7 (3.4)	8.9 (1.5)
No. regenerating stems ha^{-1}	40.0 (17.3)	106.7 (91.7)	653.3 (234.1)	0.0	3.3 (2.2)	6.7 (6.7)
No. trees ha^{-1}	146.7 (31.8)	110 (38)	143.3 (3.3)	40 (0)	56.7 (8.8)	50 (25.2)
Ratio regenerating stems/adult	0.27	0.97	4.56	0.00	0.06	0.13
Structural complexity index score	80.0	64.6	84.8	64.8	53.5	56.8

5.2.2 Establishment treatments

Within each research site an area approximately 100m x 100m (0.1ha) was identified in which sixteen 5m x 5m plots were subjectively placed in canopy gaps (> 2m from a tree trunk). Treatments were randomly allocated to plots so that there were eight plots that were cultivated and eight plots that were burnt at each site as follows:

- **cultivation**: an initial spray with glyphosate herbicide (360g/L active ingredient at 10ml/L) and subsequent cultivation with a rotary hoe to a depth of approximately 20cm once sprayed vegetation had died (persistent *Lomandra longifolia* was cut back to assist cultivation where necessary) OR
- **burning**: coarse woody debris was piled across the plot and burnt to produce an ashbed. Heavy fuels (logs and branches) were mixed with lighter fuels (sticks, bark and, at the Oatlands sites where woody debris was scarce, straw) to assist ignition and to maintain an intense burn. Evercreech coupe piles were predominantly in the form of windrows of logging debris and were twice the volume of those at other sites (Figure 5-1).

Figure 5-1 A typical spot fire for the burn treatments at Epping Forest (centre) and Evercreech coupe (right)

Loosely stacked piles of eucalypt timber in windrows weigh approximate 300kg m^{-3} (Humphreys and Craig 1981) equating to 450 tha^{-1} . With most piles at Evercreech coupe being 1.5-1.75m high, there would have been approximately 580 tha^{-1} of fuel

concentrated in the piles used. The piles at other sites were smaller (approximately 0.75m high) and more open and would thus have contained approximately 370 tha^{-1} of fuel. All piles produced peak flame heights in excess of 3m, generally indicating fires of high intensity (Morrison 2002). The piles burnt for 4-5 hours until very little woody residue remained and soils were burnt black with substantial layers of ash and charcoal. Some areas of the ashbeds at the Evercreech coupe site were burnt red (indicating extremely intense fire, Launonen *et al.*1999) and these areas were not used for experimental plots.

After the burns or cultivation were applied, two large logs (30-50cm diameter) were placed 60cm apart running east west on half of the sixteen plots. This log addition treatment was randomly allocated to four of the eight burnt plots and four of the eight cultivated plots. The remaining plots had no logs added and any coarse woody residue was removed. This resulted in four replicates of four establishment treatments at each site: burnt with logs (BL), burnt no logs (BN), cultivated with logs (CL), cultivated no logs (CN) (Figure 5-2).

The overall design was: Six experimental sites with four replicate 5m x 5m plots of each of four treatments (BL,BN,CN,CL) with four quadrats of hand sown eucalypt seed in each plot (see section 5.2.4 for seed sowing rates and Figure 6-1 in following chapter for schematic diagram of a plot).

Establishment treatments were undertaken in late autumn 2008 and not sown with seed until the following spring (see Table 5-1 for dates) to allow time for moisture to build up in the soil and to avoid frosts over winter (Davidson and Close 2006).

Prior to sowing/planting at Oatlands and immediately post sowing/planting at the other sites, all plots were fenced to eliminate grazing by domestic, native and feral

animals as a factor affecting seedling survival (Figure 5-3). At the Tom Gibson and Valleyfield sites at Epping Forest four fenced untreated, unplanted control plots were also established to test whether fencing alone promotes eucalypt regeneration.

Figure 5-2 Examples of the four establishment treatment plots: burnt with 2 large logs (BL), burnt no logs (BN), cultivated with 2 logs (CL), cultivated no with no logs added (CN).

Figure 5-3 Typical fences around plots at Epping Forest and Fingal sites (left) and at Oatlands sites (right).

At Valleyfield, Liliesleaf and Bald Hill four unfenced, untreated 5m x5m control plots were marked out inside the remnants (these remnants were surrounded by

fences that excluded livestock but not native browsers) and outside the remnants in directly adjacent grazed paddocks. The distance from the centre of each plot to the closest tree was measured at every site except the Evercreech coupe.

5.2.3 Seed collection and viability testing

Seed was collected from six parent trees at each of the three locations. *Eucalyptus amygdalina* seed was collected at Tom Gibson Reserve (Epping Forest) and Evercreech Reserve (Fingal) and *Eucalyptus pauciflora* on the Liliesleaf property at Oatlands. The seed from each tree was tested for viability in March 2008 using germination tests on four replicates of 0.4g samples for *E. amygdalina* and 0.5g samples for *E. pauciflora.* Prior to testing, the seed was wet stratified in a fridge at 4 °C for 3 weeks (*E. amygdalina*) or 6 weeks (*E. pauciflora*) as suggested by Boland *et al.* (1980), in the manner described by Langkamp (1987).

5.2.4 Seed sowing rates

Forestry technical manuals suggest that only 5% or less of fertile seeds used in broadcast field sowings will produce a seedling capable of surviving the initial establishment period (Lockett 1991). Suggested sowing rates on unfavourable (dry) coupe sites for *E. amygdalina* are 0.6 kg ha^{-1} and *E. pauciflora* 1.2 kg ha^{-1} (Lockett 1991). However for 2 m^2 indicator plots in coupes, 50 times the coupe sowing rate is sown per plot, aiming to produce about 25 well established seedlings in each plot (Neyland and Edwards 1999). The sowing rates in this study were also based on 50 times the broadcast sowing rate (0.6 kg ha^{-1} = 0.06 gm^{-2}, 0.06 x 50 = 3 gm^{-2} for *E. amygdalina* and 1.2 kg ha^{-1} = 0.12 gm^{-2}, 0.12 x 50 = 6 gm^{-2} for *E. pauciflora*). This approximately equates to 0.3 g per 30 x 30cm quadrat for *E. amygdalina* and 0.6 g per quadrat for *E. pauciflora*. However, extra *E. amygdalina* seed was available so

0.4 g per quadrat was sown. Battaglia (1996) working on *E. amygdalina* seedling germination in slightly more benign field conditions in Tasmania, used 2.5 gm^{-2} of seed, and in direct seeding trials in the Midlands Pinkard (1992) used 1.85 gm^{-2} of *E. amygdalina* and 4.7 gm^{-2} of *E. pauciflora* seed, generally consistent with the sowing rates used in this study.

Four 30 x 30 cm quadrats were marked out in a line on the western edge of all plots to assess whether aspect affects germination and survival. In plots with logs the quadrats had aspects relative to log of north, inner south (between logs), inner north (between logs) or south (see Figure 6-1 in following chapter for schematic diagram of overall plot layout). Cold stratified pre-primed eucalypt seeds (weighed then soaked in water over night prior to sowing) were sown by hand onto the pre-wet soil surface in each quadrat and then watered with a watering can.

5.2.5 Herbicide application

A single post-planting application of Eucmix® GR granular herbicide (Macspred Australia, Delacombe, Victoria, active ingredients 44 g/kg Terbacil and 2 g/kg Sulfometuron Methyl) was made in late November 2009 in the cultivated plots at the degraded sites: Evercreech plantation, Valleyfield, Liliesleaf and Bald Hill. It was not deemed necessary to treat the burnt plots at these sites as invasion by exotic species was minimal at that time whereas the cultivated plots had been invaded with pasture grasses and herbaceous weeds. No herbicide treatments were made at Tom Gibson Reserve or Evercreech Coupe as there is no exotic weed problem at these sites.

Eucmix® GR granular herbicide has an initial knockdown as well as a residual effect and was chosen because it has been shown to have little effect on eucalypts but good

control of pasture grasses (Churchill 2004). The herbicide was applied with a purpose built dispenser that distributed 3g per 1 m diameter circle according to the manufacturers recommended dose for young eucalypt plantations for the control of certain annual and perennial weeds (Churchill 2004). The application was made so that there was little contact with the eucalypt seedlings and the herbicide was spread evenly across the plot. The level of weed control achieved differed among sites.

5.2.6 Assessment

Every two weeks until 2 months after sowing, then every month up to 6 months, then at 9 months and approximately 12 months after sowing, the new germinants in sown seed quadrats were counted and marked individually with toothpicks which were colour coded for the date when first observed. For self-sown seedlings, their species and position in the plots were also recorded at the above times. Death of germinants was noted at each census date. Heights of germinated seedlings were measured once they were approximately 2.5cm or greater.

Total germination is the sum of all observed seedlings that emerged during the census period. Survival percentage is the number of germinants alive at the census date as a percentage of the total germination. Seedling establishment is the proportion of seedlings that existed at the end of the monitoring period.

Table 5-3 Length of time each experimental site was assessed for germination and survival of seedlings plus which plot level variables were measured at each site.

Site	Length of assessment	Vegetation cover	Distance from nearest tree	Water repellency (WDPT)
Tom Gibson	12 months	No	Yes	Yes
Valleyfield	22 months	Yes	yes	Yes
Liliesleaf	22 months	Yes	Yes	Yes
Bald Hill	22 months	Yes	Yes	Yes
Evercreech Plantation	12 months	No	Yes	Yes
Evercreech Coupe	4 months	No	No	Yes

The Evercreech coupe site was only assessed for germination and survival data until February 2009 (4 months after sowing) as the fencing surrounding some of the plots was stolen prior to the next census. Tom Gibson Reserve was only assessed for a year as an ecological burn was subsequently undertaken in the section of the reserve where the study plots were and the plots were differentially affected by fire (Table 5-3).

The three most degraded sites (Bald Hill, Liliesleaf and Valleyfield) were assessed again for seedling survival approximately 22 months after sowing. At this time (28 months after treatments were prepared) Domin cover scores of vegetation guilds and ground covers (as described in section 3.2.3) were estimated in the treatment plots, fenced control plots (Valleyfield) and unfenced 5m x 5m control plots (Bald Hill and Liliesleaf) inside the fenced area of the remnant (Table 5-3). Cover scores were also estimated for four unfenced 5m x 5m control plots on the outside of the remnant fences (in surrounding paddock) which were aligned with plots on the inside of the fence.

5.2.7 Soil water repellency

Five soil samples were taken from each experimental plot in August 2008 prior to planting, using a soil sampler with a 2cm diameter x 10cm long collection cylinder. These samples were bulked into a single sample for each plot, air dried and stored in plastic bags in the laboratory. Potential water repellency persistence was subsequently measured for each plot using the Water Drop Penetration Time test (WDPT) following the methods described in section 3.2.4 of this thesis. Applied water droplets were followed for up to 3 hours. Soil from each plot was allocated a water repellency class as described in Table 3.2.

5.2.8 Data Analysis

Counts of observed germination and seedling survival from sown seed were summed to plot level (representing the sum of the four sown quadrats in each plot). When data from five sites (not including Evercreech coupe) were analysed the sown and self-sown seed data had Poisson distributions with overdispersion. Power transformations did not improve normality of the data. Therefore rank based non-parametric Kruskal Wallis tests were undertaken to determine whether total observed germination of manually sown and of self-sown seed (cumulative over a year) and seedling survival (at the end of the year) were significantly different among sites from all locations and establishment treatments.

Sown seed data from the two Epping Forest sites alone were normalized by log transformations and thus ANOVA was used to test whether there was a significant difference in germination and survival of sown *Eucalyptus amygdalina* seed between a healthy (Tom Gibson) and degraded (Valleyfield) site and among treatments at these sites. Data for self-sown seed was unable to be normalized by transformation and thus Kruskal Wallis tests were used to check for significant differences in germination and survival over a year between the two sites.

Kruskal Wallis tests were also used to test whether there were differences in observed germination and survival respectively among treatments within each site over a year (plus Evercreech coupe at four months) and whether there were differences in seedling survival among sites and among treatments nearly two years after sowing at the three degraded sites assessed. A Kruskal Wallis test was undertaken on quadrat level data of plots with logs from five sites to test whether there was an effect of quadrat aspect on germination and survival of sown seed.

Spearman's rank correlation tests were undertaken to determine whether the germination of seed and survival of seedlings established from seed at the six research sites were associated with structural complexity index scores or individual structural attributes of the sites (listed in Table 5-2).

The numbers of plots in each establishment treatment that did or did not contain surviving seedlings (sown seed and self-sown combined) were counted at five sites at the one year after sowing census and at Liliesleaf, Bald Hill and Valleyfield after 2 years (Table 5-3). A Fisher's exact test of independence was applied to determine if sites significantly differed from each other in the number of plots that contained live seedlings at the end of one and two years respectively. A Fisher's exact test of independence was also applied to end of second year data to determine if the number of plots that contained live seedlings significantly differed between treatments and untreated controls.

Domin cover data were checked for normality of distribution and transformed where necessary (grass cover square root, herb and graminoid cover log transformed). ANOVA were run on data that had normal distributions after transformation (as determined by Shapiro Wilks tests in R). Where transformations were unable to improve normality (mainly due to high numbers of zero scores), non-parametric Kruskal Wallis tests were run to determine whether there were differences in the cover scores among treatments. Spearman's rank correlation tests were undertaken to determine whether the numbers of seedlings alive in a plot two years after sowing (sown and self-sown combined) were associated with vegetation or ground cover scores.

Kruskal Wallis non-parametric tests were undertaken on plot water repellency (WDPT) data to determine whether there were differences in soil water repellency among sites, treatments and burnt and cultivated plots. Spearman's rank correlation tests were undertaken to determine whether the total observed germination of seed and survival of seedlings established from seed a year after sowing were associated with soil water repellency.

Generalised linear modelling with a binomial logit link (logistic regression) was undertaken on treated plot data (not controls) to determine whether the probability of a plot containing seedlings (sown and self-sown combined) a year after sowing could be predicted by site, treatment, distance from nearest tree or water repellency (WDPT) and after two years (for the three sites assessed) for these same variables plus the cover of vegetation and ground covers. The generalised Wald statistic (Z) with a χ^2 distribution was used ("glm" function in R) to test whether the coefficient of a predictor was significantly different to zero and thus should be included in the model (Quinn and Keough 2002).

5.3 Results

5.3.1 Seed viability

The results from the germination tests for Epping Forest *E. amygdalina* were 175 ± 40 germinants per gram, Evercreech *E. amygdalina* 349 ± 41 germinants per gram and Oatlands *E. pauciflora* 91±15 germinants per gram which exceeded those reported by Boland *et al.* (1980) (*E. amygdalina* 137 ± 121 and *E. pauciflora* 59 ± 23 germinants per gram). At the sowing rates of 0.4 g for *E. amygdalina* and 0.6 g *E. pauciflora* there would have been approximately 70 viable seeds sown in each 30 x 30 cm quadrat at Epping Forest, 140 at Evercreech and 55 at Oatlands.

5.3.2 Year 1 germination and survival

Sown seed: There was a significant difference among sites ($H_5 = 17.73$ $p = 0.001$) in the amount of observed germination in sown plots (Table 5-4). The healthy Tom Gibson Reserve site had a significantly greater number of germinants than each of the degraded sites, which were not significantly different from each other (Table 5-4). However the number of germinated seedlings that were still alive at the one year census was not significantly different among sites ($H_5 = 5.14$ $p = 0.27$). This result would have been influenced by the fact that many of the seedlings alive at the year census at the degraded sites (other than Valleyfield) were new recruits since the previous census, essentially representing a new cohort and thus boosting survival rates at the time (Table 5-4).

When the Epping Forest sites data were analysed with ANOVA there was a significant difference in sown seed germination ($F_{1,24} = 10.46$, $p= 0.003$) and survival ($F_{1,24} = 6.16$, $p= 0.02$) with Tom Gibson having significantly more of both than Valleyfield.

Table 5-4 Number of observed germinants and survivors and resulting survival percentage for seed sown in quadrats and seed that was self-sown in plots with four establishment treatments at six research sites. Dates of the last census date (i.e. the date survivors were last observed) are given in brackets for each site and are approximately one year after sowing seed except Evercreech Coupe which was only monitored for four months after sowing. * indicates that survivors include new germinants first seen at last census.

		Sown seed in quadrats			Self-sown seed		
Site, species (last census)	**Treatment**	Observed Germinants	Survivors	Survival (%)	Observed Germinants	Survivors	Survival (%)
Tom Gibson	Burn with log	59	3	5.1	65	17*	26.1
E.amygdalina	Burn no log	62	3	4.8	47	18*	38.3
(17/9/2009)	Cultivated with log	110	12	10.9	1	0	0
	Cultivated no log	182	18*	9.9	10	0	0
	Total	**413**	**36***	**8.7**	**123**	**35***	**28.4**
Valleyfield	Burn with log	9	3*	33.3	6	2*	33.3
E.amygdalina	Burn no log	5	1	20	5	0	0
(24/9/2009)	Cultivated with log	31	0	0	6	0	0
	Cultivated no log	22	3	13.6	4	2	50
	Total	**67**	**7***	**11.1**	**21**	**4***	**19**
Evercreech	Burn with log	0	0	0	0	0	0
Coupe	Burn no log	0	0	0	0	0	0
E.amygdalina	Cultivated with log	72	50*	69.4	12	12	100
(2/2/2009)	Cultivated no log	89	61*	68.5	3	3	100
	Total	**161**	**111***	**68.9**	**15**	**15**	**100**
Evercreech	Burn with log	29	12*	41	0	0	0
Plantation	Burn no log	15	15*	100	110	110*	100
E.amygdalina	Cultivated with log	4	4*	100	1	1*	100
(29/10/2009)	Cultivated no log	2	0	0	1	1*	100
	Total	**50**	**31***	**62**	**112**	**112***	**100***
Liliesleaf	Burn with log	20	20*	100	15	4	26.7
E.pauciflora	Burn no log	3	3*	100	5	3	60
(12/11/2009)	Cultivated with log	18	18*	100	4	3*	75
	Cultivated no log	6	6*	100	1	1	100
	Total	**47**	**47***	**100**	**25**	**21***	**84**
Bald Hill	Burn with log	5	5*	100	1	1	100
E.pauciflora	Burn no log	1	1	100	5	5*	100
(12/11/2009)	Cultivated with log	15	13*	86.7	5	5	100
	Cultivated no log	10	10*	100	0	0	0
	Total	**31**	**29***	**93.5**	**11**	**11***	**100**

The estimated amount of viable seed sown at each site (16 plots x 4 quadrats x number of estimated seed sown per quadrat), the observed germination percentage of viable seed and the percentage of estimated viable seed that produced a live seedling after a year for five sites and over 4 months for Evercreech coupe are shown in Table 5-5.

Table 5-5 Estimated viable seed sown and germination and establishment percentages of estimated viable seed sown by the end of one year after sowing at five sites and after 4 months at Evercreech Coupe.

Site	Estimated viable seed sown	Germination percentage	Establishment percentage
Tom Gibson	4480	9.2	0.80
Valleyfield	4480	1.4	0.16
Liliesleaf	3520	1.3	1.33
Bald Hill	3520	0.9	0.82
Evercreech Plantation	8960	0.6	0.35
Evercreech Coupe	8960	1.8	na

Self-sown: Seed that germinated outside the sown quadrat areas was assumed to be recruitment from natural seed rain (self-sown). It is recognised that it is possible that some added seed could have been moved from quadrat areas by ants, wind or rain (Clarke 2000). However this is impossible to quantify and sown seeds were well watered in with most observed to settle into dips and cracks in the soil. The majority of seedlings that emerged outside quadrats were at some distance from them. The number of observed self-sown germinants did not significantly differ among sites (H_5= 8.04, p = 0.09) nor did those that were alive at the one year census (H_5=4.72, p = 0.32).

The numbers of germinants observed from self-sown seed at the Epping Forest sites were approximately a third of the sown seed germinants. However, survival rates were much greater resulting in similar number of survivors overall (Table 5-4). The Evercreech sites had a hundred percent survival of observed self-sown seedlings.

However, a large proportion of the seedlings alive (both sown and self-sown) at the Evercreech Plantation were new recruits since the previous census in March 2009 (when there was only 8 seedlings in sown quadrats and 1 self-sown seedling surviving) and thus the survival rate would be expected to decrease over time. All of the self-sown seedlings at Bald Hill survived with few new recruits seen at the one year census and only four died at Liliesleaf (Figure 5-4).

There was no eucalypt germination observed in the fenced untreated control plots at Tom Gibson and Valleyfield over the first year.

5.3.3 Structural complexity

The total number of observed germinants (sown and self-sown seed combined) at each site was significantly and strongly correlated with structural complexity index score ($r = 0.942$, $p= 0.017$) but the number of overall survivors was not ($r = 0.657$, $p= 0.175$). There was also a strong and significant correlation between the quantity of observed germination and the total length of logs at a site ($r= 0.886$, $p=0.033$). No other structural variable measured for the structural complexity index was correlated with germination and no structural attributes were significantly correlated with the number of seedlings established from seed and surviving after a year.

5.3.4 Establishment treatment effects

There was a marginally significant effect of establishment treatment on sown seed germination across all five sites ($H_3=7.79$, $p= 0.05$) with their being more germination in the cultivated with log plots (average 8.6 ± 2.7) than the burn with no log treatment (average 4.1 ± 2.3). No other combination of treatments was significantly different from any other and germination was very variable among

replicate plots within treatments. In the plots that had logs added there was no effect of quadrat aspect on germination (H_3=1.51, p= 0.68) or survival (H_3=0.78, p= 0.85).

There was no germination at all in the burnt plots at Evercreech Coupe (Table 5-4). The ashbeds produced at the coupe site were much thicker than at other sites due to the greater amount of fuel in the spot fires and self-sown seedlings were observed to germinate only on the edges of the ashbeds (outside the experimental plots) where regeneration was prolific.

There was no significant differences in sown seedling survival among treatments (H_3= 2.62, p=0.45) across all sites (with Evercreech coupe excluded), with different treatments producing the most survivors at nearly every site respectively (Table 5-4).

The amount of observed germination of self-sown seedlings (H_3=7.22 p = 0.07) and surviving self-sown seedlings at the one year census (H_3= 6.63, p = 0.08) also did not significantly differ across establishment treatments.

At the end of one year's observation at the five sites, 10 burn with log plots, 16 burn no log plots, 12 cultivated with log plots and 11 cultivated with no log plots contained seedlings all of which were significantly more than the fenced control plots which all had no seedlings ($p < 0.001$). This left 31 plots, or 38.75% of treated plots without seedlings one year after sowing. Tom Gibson and Liliesleaf had significantly more plots with live seedlings at the year one census than Valleyfield (p=0.028) but all other combinations of sites had similar numbers (half or more) of plots with seedlings (Table 5-6).

Table 5-6 Number of 5mx5m treatment plots (out of 16) with live sown or self-sown seedlings and the total number of plots with live seedlings (sown and self-sown combined) at first and second year censuses.

Site	Year 1 census			Year 2 census		
	No of plots with live sown seedlings	No of plots with live self-sown seedlings	Total no of plots with live seedlings	No of plots with live sown seedlings	No of plots with live self-sown seedlings	Total no of plots with live seedlings
Ev. Plantation	5	4	8	not assessed	not assessed	not assessed
Tom Gibson	10	6	13	not assessed	not assessed	not assessed
Valleyfield	5	2	5	4	2	4
Liliesleaf	8	6	12	1	3	4
Bald Hill	7	7	11	9	9	11

5.3.5 Year 2 survival (3 sites)

Germination in manually sown plots peaked in the first few months after sowing at Valleyfield but not until late spring during the year after sowing at Bald Hill and Liliesleaf at Oatlands (Figure 5-4).

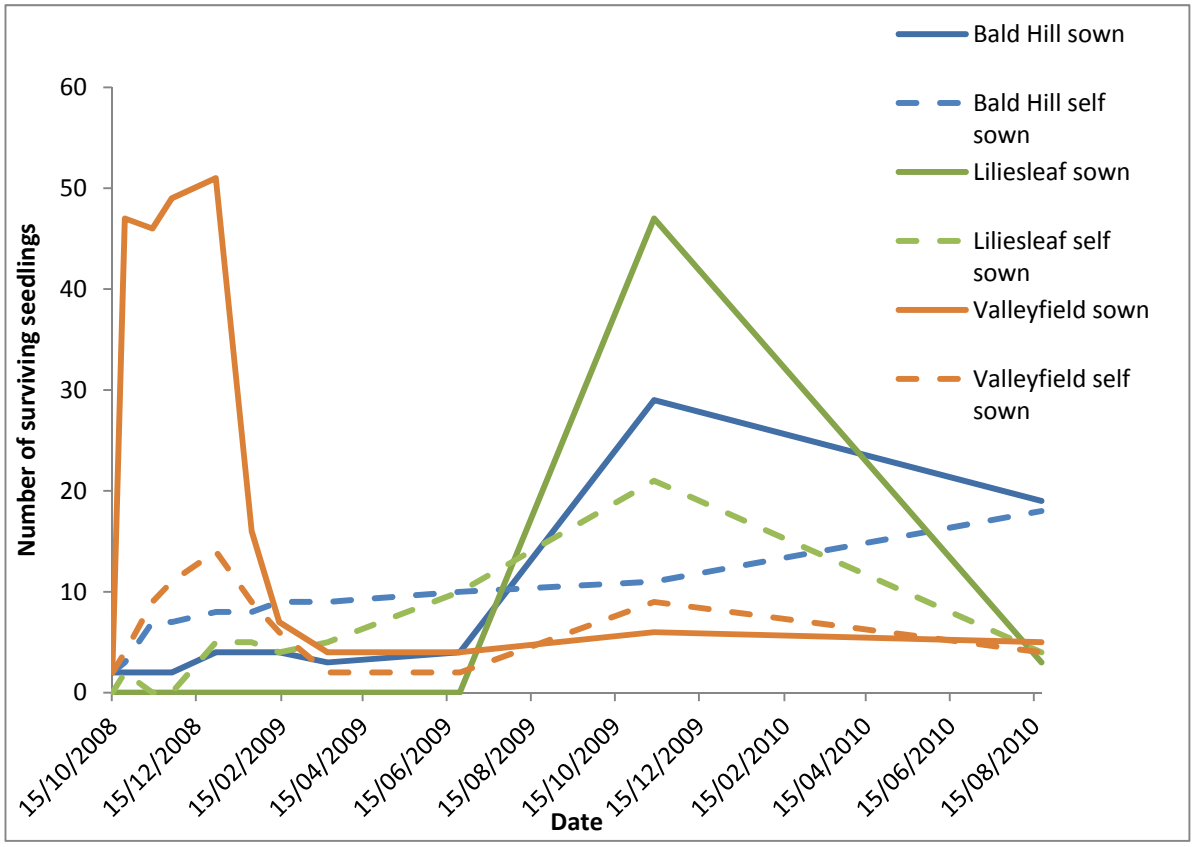

Figure 5-4 Number of living seedlings germinated from seed over a 22 month period in manually sown quadrats and outside sown quadrats (self-sown) at three Midlands sites: Bald Hill, Liliesleaf at Oatlands, Valleyfield at Epping Forest.

Bald Hill had the greatest number of plots with seedlings (Fisher's Exact Test p value = 0.021, Table 5-6) and the greatest amount of seedlings surviving at the 22 month census ($H_2 = 7.69$, p= 0.02) with almost 4 times more live seedlings than at Liliesleaf and Valleyfield. Only 3 of the 47 new recruits in manually sown plots observed in the one year census at Liliesleaf were still alive at the 22 month census (Figure 5-4), all of which were in one cultivated with log plot.

Second year survival rates of self-sown seedlings were more than double the survival rates of germinants from manually sown seed (Table 5-7).

Table 5-7 Estimated survival percentage of manually sown seed and survival percentage of observed sown germinants and observed self-sown germinants at the 22 month census at three sites

Site	Survival % of estimated viable seed sown	Survival % of observed sown germinants	Survival % of observed self-sown germinants
Valleyfield	0.11	8.6	18.8
Liliesleaf	0.08	6.4	16
Bald Hill	0.57	48.7	89.5

While there was more than twice as many seedlings in cultivated with log plots than any other treatments, the variability among sites and among plots within sites meant this was not statistically significant ($H_3 = 0.54$, p = 0.91). Overall manually sown seed and self-sown seed produced comparable numbers of survivors after 2 years of observation at these three degraded sites, thus sowing seed effectively doubled the number of seedlings that established over the first two years.

No eucalypt seedlings were observed to germinate in the fenced untreated controls at both Epping Forest sites over the two years of monitoring. The Fisher's exact count test of independence showed that all establishment treatments resulted in

significantly more plots with seedlings compared to the controls (p= 0.025) but were not significantly different from each other.

The average distance of a seedling from an added log in the plots with logs was 21.6 ± 5.5 cm. The most common aspect of seedlings was on the southern side of logs (18 seedlings), then inner north (8 seedlings) with north and inner south aspects having 3 seedlings each.

5.3.6 Vegetation and ground cover after 2 years

Correlation with number of seedlings

There were significant positive correlations among the number of surviving seedlings in a plot and the extent of bare earth (r= 0.31, p= 0.007), cover of charcoal (r=0.294, p= 0.012) and tall shrubs (r=0.294, p=0.012) and negative correlations between the number of seedlings in a plot and the cover of herbs (r= -0.247, p=0.036) and moss (r= -0.251, p=0.033). No other vegetation or ground covers were significantly correlated with seedling numbers (results not shown).

Bare Earth

The control plots inside the remnants (0.75 ± 0.2 %) and outside the fences (1.1 ± 0.3 %) had significantly less bare earth (H_6= 31.8027 p<0.001) than all the establishment treatment plots (BL 7.9 ± 2.7, BN 11.6 ± 3.4, CL 7.5 ± 1.9, CN 9.7± 3.6 %) and Valleyfield had significantly less bare earth than Liliesleaf but not Bald Hill (H_2= 8.3, p= 0.01).

Herbs

There was a significant interaction among site and treatment in the cover of herbs (predominately exotic weed species) 28 months after plot preparation ($F_{10,54}$ =5.88,

p<0.001). At Bald Hill cover of exotic herbs was similar and relatively low in all establishment treatments (BL 2.5 ± 0, BN 2.7 ±1.6, CL 2 ± 0.5, CN 3.2 ± 1.5 %) compared to the controls inside the remnant woodland (41.25 ± 9.2%). At Liliesleaf the two cultivated treatments (CL 18 ± 0, CN 21.4 ± 7.3 %) had significantly more exotic herbs than the burnt with log plots (5 ± 1.4 %) and the inner controls (3.2 ± 1.5 %) but all others were not significantly different. At Valleyfield the burnt with no log treatment had significantly less exotic herb cover (1 ± 0.5 %) than the cultivated treatments (CL 25.9 ± 8.3, CN 25.9 ± 8.3 %) but none was significantly different to the controls.

Grass

There were significant site ($F_{2,56}$ = 6.69 , p= 0.002) and treatment ($F_{6,56}$= 7.71, p<0.001) effects on the cover of grass at the 3 most degraded sites approximately 28 months after preparation. Overall grass cover was significantly less at Liliesleaf at Oatlands than both Valleyfield and Bald Hill which were not different to each other. All the burnt and cultivated treatments (BL 25.7 ± 7.6, BN 13.6 ± 5.1, CL 29.6 ± 6.5, CN 31.6 ± 7.5 %) produced significantly less grass cover than the plots in the surrounding paddocks (69.9 ± 6.6 %) but only the burnt no log plots had significantly less grass cover than the control plots in the remnant woodland (39.7 ± 9.6 %).

Graminoids

There was a significant interaction between site and treatment ($F_{10, 54}$ = 8.25 p<0.001) in the cover of graminoids (predominantly *Lomandra longifolia*). There was significantly more *Lomandra* cover in the controls inside the woodland remnant at Bald Hill (41.25 ± 9.9 %) than in each of the establishment treatments (BL 3.7 ± 1.2, BN 2.7 ± 1.6, CL 2 ± 0.5 and CN 3.2 ± 1.5%). At Liliesleaf there was no *Lomandra* in the surrounding paddocks and thus all treatments in the remnant had

significantly more *Lomandra* cover than outside. The inner controls (56.7 ± 13.7 %) had significantly greater cover than the burnt with no log treatment (3.9 ± 2.1 %) but all other treatment combinations were not significantly different from each other. At Valleyfield there was low cover of *Lomandra* in all the plots and controls within the remnant woodland but significantly more cover in the surrounding paddock (5.9 ± 4.1 %).

Moss

There was less moss in burnt plots than in all other plots, for all sites (H_5=28.5, $p<0.001$).

There were significant differences among sites in the cover of ferns (Bald Hill and Liliesleaf > Valleyfield), tall shrubs (Bald Hill and Valleyfield > Liliesleaf) and litter (Bald Hill and Liliesleaf > Valleyfield), but not among treatments (data not shown).

5.3.7 Soil water repellency

Water repellency (as measured by WDPT) was significantly different among sites (H_5=44.26, $p<0.001$, Table 5-8) but not among treatments (H_3= 0.60, p= 0.89) or between burnt and unburnt plots (H_1= 0.015 p= 0.90). There was no correlation between plot level soil water repellency and seed germination (r=0.06, p=0.553) or survival of resulting seedlings (r=-0.02 p=0.803), respectively.

Table 5-8 Median soil water repellency persistence as measured by water drop penetration time (WDPT), significant differences among sites (represented by different letters) and descriptive water repellence category (refer to Table 3-2) of median WDPT class of 16 plots at each of 6 research sites

Site	Median WDPT class	Significant difference	Water repellence category
Tom Gibson Reserve	5	ac	extreme
Valleyfield	5.5	a	extreme
Evercreech coup	2	b	low
Evercreech plantation	4.5	ac	severe/extreme
Liliesleaf	3	c	moderate
Bald Hill	6	a	extreme

5.3.8 Probability of a plot containing seedlings

The only significant predictor of a plot containing at least one surviving seedling was research site, both at one year ($Z_{4,75}$ = -2.067 p = 0.039) and 2 years ($Z_{3,45}$ = -2.388 p = 0.017) following sowing. At one year, plots in Valleyfield had the lowest likelihood of having newly recruited live seedlings. At two years, plots at Bald Hill had a greater probability of having a surviving seedling than at either Valleyfield or Liliesleaf.

5.4 Discussion

5.4.1 Recreating the recruitment niche

This study confirmed findings by others (McCormick 1991; Pinkard 1992; Stoneman 1994; Yates *et al.* 1996; Clarke 2002) that the germination niche of woodland eucalypts is much broader than the establishment niche showing that conditions for establishment are more stringent than conditions for germination (Turnbull *et al.* 2000). A favourable microsite for germination was not necessarily a favourable site

for seedling survival (Battaglia and Reid 1993; Schupp 1995) as many more seeds germinated than survived during the study period both in healthy and degraded sites.

Establishment treatments of burning or cultivation did each significantly increase the number of plots with eucalypt recruitment in degraded remnants two years after treatment compared to fenced and unfenced untreated control plots that had no observed germination at all over the two years. There was also a significant positive association between the amount of established seedlings in a plot and the extent of bare earth and a negative association with the cover of exotic herbs. These results confirm that lack of seed bed condition (through lack of soil disturbance and presence of competitors) is a major barrier to eucalypt recruitment in degraded woodlands (Fensham and Kirkpatrick 1992; Yates *et al.* 1996; Semple and Koen 1997) and supports the proposition that some sort of soil amelioration that mimics large scale disturbance plus weed control are required to promote recruitment from seed (McCormick 1991; Orr and Todd 1992; Stoneman 1994; Yates *et al.* 1994b; Yates *et al.* 2000a).

The designated analogue to the natural recruitment niche was burn with log treatment; however this did not significantly improve recruitment compared to other treatments. There was little consistency in the treatment effects on survival of seedlings from manually sown and self-sown seed. This was the case across all research sites a year after sowing and within the three degraded sites measured after two years. Ruthroff et al (2010) also found no differences in survival of seedlings in ashbed and ripping treatments in a multi species seeding trial (which included eucalypts) in degraded Western Australia Tuart woodlands.

Neither burning nor addition of logs significantly increased the probability of a treated plot having seedlings established at one or two years after sowing compared to cultivation. There were very similar numbers of plots in each of the four treatments (burnt with or without logs and cultivation with or without logs) that contained established seedlings at both one and two years suggesting that the recruitment niche can actually be broadened through the use of cultivation as an alternative restoration technique to fire. This is in agreement with Tasmania forestry manuals for managing dry production forests that suggest that fire (through slash burning) is not always necessary to induce eucalypt recruitment and may not even be desirable when the protection of existing trees and regeneration is important (Lockett and Candy 1984; McCormick 1991; Orr and Todd 1992; ForestryCommission 1993).

McCormick (1991) suggests scarification (exposure of surface mineral soil by mechanical removal of vegetation and litter layers) as an alternative to fire. Scarification aims at exposure with minimal soil disturbance and thus may not be as suitable as cultivation at sites where soil compaction is a possible barrier to recruitment. At both healthy sites in the current study there was significantly more recruitment from manually sown seed in cultivated plots than in uncultivated plots. However across all sites, while there was significantly more germination in cultivated with log plots than burnt with no log plots, there was no significant difference in survival between these treatments after one and two years.

5.4.2 Overcoming barriers to recruitment

Competition removal

One of the main apparent benefits of the intense fire treatment, compared to cultivation, was the longer term weed control it offered without the addition of

herbicide. Forestry manuals (McCormick 1991; Orr and Todd 1992) propose that areas of intense burn are not readily reinvaded with grass while areas of ground disturbance and/or low intensity burn can be quickly invaded by grass in healthy dry forests with a grassy understorey. This was also shown to be the case in remnants degraded by grazing in the current study as cultivated treatments had similar levels of grass as untreated controls two years after treatment while burnt plots without logs had significantly less grass at all three sites assessed. Burnt plots with logs also had similar levels of grass to untreated controls, and this may be because the logs could act as obstacles to, and lodgement sites for, windblown grass seed (Lamont *et al.* 1993; Ludwig and Tongway 1996) and thus be more readily invaded than burnt plots without logs.

More than two years after treatment, burnt with no log plots also maintained more bare earth and a lower cover of *Lomandra* and herbaceous weeds at most sites compared to untreated control plots within the woodland remnants. At two sites, the burnt plots had significantly less cover of herbaceous weeds than cultivated plots or controls, despite granular herbicide application on cultivated plots in the first year. At Bald Hill, the degraded site in which seedling recruitment was highest, all treatment plots maintained a significantly lower level of exotic weed species and *Lomandra* compared to controls showing that the success of weed control is site specific.

The critical need for good weed control for at least the first few years following introduced sowing (Pinkard 1992; Yates and Hobbs 1997a; Yates *et al.* 2000a) was emphasised by the negative correlation between the number of eucalypt seedlings established in a plot and cover of exotic herbs, and this was reinforced by the positive correlation of seedling establishment and the extent of bare earth. The

competitive effects of weeds that developed over time in cultivated plots may explain why significantly better germination in cultivated plots did not subsequently result in improved or sustained recruitment compared to burnt plots. Possible adverse effects of the granular herbicide on recently germinated seedlings may also have contributed to this result.

Pinkard (1992) demonstrated the importance of long term weed control and the difficulty in achieving it in work on direct seeding of tree species in pasture sites in the Midlands of Tasmania. She found pre-sowing knockdown herbicide gave relatively short periods of weed control, suggesting the need for follow up applications, as was the case in this study. Applications of pre-emergent herbicides and scalping of the grass sward were other weed control measures trialled by Pinkard (1992) which gave more effective control but did not increase eucalypt seedling emergence or survival. More effective weed control in the current study may have been achieved in cultivated plots with a pre-sowing application of herbicide (in addition to initial glyphosate application prior to cultivation) rather than, or as well as, post-planting herbicide application (Florence 1996; Close and Davidson 2003; Nardon *et al.* 2005). This would be a strong recommendation in future restoration efforts where the use of fire as an establishment treatment is not possible.

Weed control was not undertaken at the healthy Tom Gibson reserve site as there were few exotic species present. However cultivated plots at this site were re invaded by native species (mainly small shrubs and grasses) as generally predicted by Orr and Todd (1992) and McCormick (1991). It is quite possible that the competitive effects from these other species, especially for water during dry summer months, could have caused the large crash in survival of eucalypt germinants in cultivated plots observed at this site. Seedlings planted in these plots also showed

remarkably reduced growth compared to those in burnt plots most likely due to competition for resources from other species (Chapter 6 results).

Spot intense burns may be the most economical and broadly effective way of achieving control over recalcitrant native and herbaceous exotic species (which otherwise represent a barrier to eucalypt recruitment) without the need for repeated follow-up control over the first year or two while eucalypt seedlings establish. However, this study also indicated that such intense burns may also have some negative consequences for eucalypt recruitment as germination was generally lower in burnt than cultivated plots. This may be related to the amount of fuel burnt, the amount of ash produced, the heterogeneity of fire intensity (or lack of it) and subsequent effects on soil properties and water availability.

Fire effects

At the Evercreech coupe site, where much greater amounts of woody fuel were burnt in each spot fire, the residence time of the fire and hence high surface temperature was prolonged and thick layers of ash developed. There was no observed germination of sown eucalypt seed in the burnt plots at this site. This was despite planted seedlings in the same plots surviving and growing particularly well. This experiment is described in Chapter 6. Recruitment from natural seed rain was observed on the margins of the ash beds (where the fuels and residual ash were lighter) and elsewhere on the site. Seeds sown into thick layers of ash may not make contact with the soil surface and therefore desiccate prior to germination (ForestryCommission 1993). This suggests that there is a threshold of fuel and ash amounts that prevents germination and establishment. However it is unlikely that this amount of fuel would be available for restoration treatments in degraded woodlands, especially where coarse woody debris loads are reduced through firewood collection.

The amount of fuel that was used in the wood piles at sites other than Evercreech coupe, while considerably less, may still have been more than was necessary to promote recruitment. The effects of fire on soil properties are also influenced by the intensity and duration of heating (Chandler *et al.* 1991; Burrows 1999; Certini 2005). The method of having spot burns with coarse woody debris piled up to a greater extent than in a natural wildfire situation may have affected microbial associations and the heterogeneity of soil physical properties such as water repellency which are important in the eucalypt recruitment niche.

Soil water repellency

As shown in Chapter 4, water repellency is highly variable vertically and horizontally in dry eucalypt forest soils. It is to be expected that this small scale heterogeneity exerts an influence on the where a seed may germinate and establish successfully. The wood piles used in this study would have resulted in fires of relatively uniform intensity over a broad (5m x 5m) area. This is in contrast to intense fire concentrated around individual pieces of coarse woody debris on the forest floor and surrounded by less intense fire in light woody fuels and litter. Spot fire treatments are likely to have resulted in a fairly uniform layer of wettable surface soil with a strongly hydrophobic layer just a few centimetres below as described in Chapter 4. When seeds were being watered after sowing, the added water was observed to infiltrate much faster in burnt plots than in cultivated plots (where the water sat in small scale dips in the cultivated soil and infiltrated very slowly). This is likely due to the water added to burnt plots flowing across the plot in the wettable layer rather than infiltrating through the hydrophobic layer beneath. This phenomenon was discussed in Chapter 4 as distribution flow and may have been even more pronounced if plots were on a slope. This phenomenon is also widely

reported in literature on run off and erosion induced by post fire water repellency (DeBano 2000; Letey 2001; Doerr *et al.* 2006; Shakesby and Doerr 2006). The lack of heterogeneity of water repellency and resultant lack of small scale sumps for water may help explain why early germination in burnt plots was significantly less than in cultivated plots.

Moderate cultivation of compacted soils increases surface soil roughness so that additional micro-catchments are made (Yates *et al.* 2000a). In hydrophobic soils, rainfall and added water could pool and slowly infiltrate in these microsites, temporarily relieving water repellency and providing moisture and humidity for seed germination (Battaglia and Reid 1993). Further research is needed to investigate these effects, however it can be expected that small scale effects such as these would have played a role in the improved germination of sown seed in cultivated plots (in healthy and degraded sites) compared to burnt ones.

No significant differences in soil water repellency were found in this study between burnt and cultivated plots and this could be due to the method of soil collection for this study (bulking 5cm deep samples) which mixed soil layers and would have given a broad estimate of average water repellency across each plot. There was also no correlation between the broad measure of water repellency and germination and survival of sown seed. The sampling in this experiment was done prior to the more detailed study described in Chapter 4. Soil sampling was not undertaken at a fine enough scale to detect the subtle differences in water repellency that influence seed germination and seedling establishment, including the layering of wettable and hydrophobic soil in burnt plots. However, the sampling did show that at most sites the soil was intrinsically and severely water repellent when dry (except for Evercreech coupe [low repellency] and Liliesleaf [moderate repellency]) which has

implications for seedling survival as soils dry out during summer and early autumn (the driest period of the year in the Midlands see Figure 6-1).

Soil water repellency is affected by soil moisture status (Leighton-Boyce *et al.* 2005; Howell *et al.* 2006). Seedlings that establish in temporarily wettable and humid microsites in cultivated soil may desiccate as the intrinsically hydrophobic soil dries out and becomes difficult to rewet. Alternatively, seedlings that have established in microsites in surface soils of burnt plots that are intrinsically wettable may be able to make better use of limited rainfall and thus survive for longer. As a seedling grows its roots explore the soil at greater depth and it becomes less sensitive to moisture deficit and other microclimatic conditions at the soil surface (Battaglia and Reid 1993; Skinner *et al.* 2009) and more responsive to conditions at depth. Even when seeds germinate in wettable surface patches, if soil lower down the profile is intrinsically hydrophobic and the seedling is not near a macropore or a vertical finger of wettable soil (see Chapter 4), the water repellency of the hydrophobic zone may cut the seedling off from longer term soil moisture stores and cause seedling death to occur well beyond the time it first emerges (Madsen 2010).

Soils that dry to become severely to extremely water repellent during summer are likely to be a strong barrier to the critical transition from germinant to established seedling. Further small scale sampling is needed to confirm this hypothesis. Trials that include soil wetting agents or gels that break down soil repellency, like those applied in other ecosystems (Osborn *et al.* 1967; Madsen 2010; Ruthrof *et al.* 2010), may also be useful in identifying methods to remove this barrier to recruitment.

Soil compaction

In Chapter 3 microsites that contained seedlings and lignotuberous sprouts had significantly softer soil than the surrounding forest floor. While it is possible that the plants themselves induce some microsite softness in the soil through physical penetration, water interception or increased organic material, intrinsically softer soil would aid root penetration depth (Skinner *et al.* 2009) and primary root growth (Passioura 1991; Misra and Gibbons 1996). In degraded remnants, soil compaction caused by heavy grazing increases the hardness of the forest floor (Yates *et al.* 2000b; Close *et al.* 2008) and limits root penetration even further. While cultivation reduces compaction and soil hardness (Passioura 1991), intense fire may increase bulk density of soil due to combustion of organic matter (Stoof *et al.* 2010). It is possible that one reason why ashbeds in this study did not significantly increase germination and survival of eucalypt seedlings compared to cultivated plots was because the fire treatment did not alleviate soil compaction or produce enough micro-scale soil roughness. A fully factorial experiment in which a cultivation plus burn treatment is included may elucidate this issue.

Coarse woody debris

In the present study there were no additive or negative effects on seedling survival with the presence of added logs on the plots. This may be due to the restoration methods used negating some of the possible benefits that coarse woody debris provides in a more natural setting (such as that described in Chapter 3). It is possible that logs and branches on the forest floor act as a browsing refuge for nearby seedlings, or reduce their apparency or exposure to mammalian browsers (Milchunas and Noy-Meir 2002; Pietrzykowski *et al.* 2003; de Chantal and Granstrom 2007) and thus have a greater probability of escaping herbivory than seedlings not protected by

coarse woody debris (Orr and Todd 1992). By placing fences around all plots in this study, the benefit of protection from browsing is also extended to seedlings not near logs. Ideally this would be tested by a fully factorial experiment with and without fences around plots but this was beyond the resources of the current study.

The placement of logs on plots after soil preparation by fire or cultivation may not have allowed sufficient time for moisture to accumulate in and under the logs prior to seeding, negating the potential benefit of coarse woody debris as a soil moisture store for nearby establishing seedlings (as displayed in Chapter 4). It is likely to take multiple rainfall events and more intimate connection of decaying logs and surface soil to build up a substantial store of moisture under logs placed on soil in essentially dry conditions (Amaranthus *et al.* 1989; Harmon *et al.* 2004). Additionally, in natural settings logs may act as lodgement sites for eucalypt seeds that are moved across the landscape by wind and water and thus greater numbers of seed may congregate and germinate near coarse woody debris than in a more open less obstructed area (Lamont *et al.* 1993; Ludwig and Tongway 1996; Semple and Koen 1997; Howell *et al.* 2006). This function was partially negated in this study by the strategic placement of seed in all plots.

The total length of logs at a site was the only structural attribute measured that was correlated with the quantity of observed germination in the plots. This may have been due to the microclimate changes that coarse woody debris can create on the surface of the forest floor. As discussed in Chapter 3, coarse woody debris can ameliorate environmental extremes, increase humidity at ground level (Harmon *et al.* 2004; Castro *et al.* 2011) and provide shelter from wind up to seven times its height on the lee side and within one times it height on the windward side (Bird *et al.* 2007). Greater amounts of coarse woody debris could provide a more benign environment

across a site at the ground level where germination occurs. The greater the amount of coarse woody debris on the ground the greater the area provided with protection from extremes. However, in this study the amount of germinants still surviving at the one-year census was not correlated with the amount of coarse woody debris at a site, showing that other factors are involved in the transition from germination to establishment.

In hindsight it would probably have been better to partially burn large logs *in situ* across plots (rather than piled up) and leave the residue behind (rather than add logs after burning treatment) to more closely mimic the natural heterogeneity described in Chapters 3 and 4. Experiments aimed at establishing the ideal amount and distribution of fuel (and thus fire intensity patterns) to best induce eucalypt recruitment could be a focus of future research. However, there may be a trade off in the effectiveness of fire on longer term weed control in degraded sites if less uniformly intense fire is used.

Seed limitation

There was at least some germination observed from natural seed rain at every site (even those of intermediate health with adult trees displaying tree decline), suggesting that lack of viable seed is not a factor that excludes eucalypt recruitment at these sites. However the addition of seed did more than double the effective density of established seedlings at the three sites which were followed for two years, indicating that seed availability may limit the amount of eucalypt recruitment at these sites (Turnbull *et al.* 2000).

There was a strong tendency for a greater percentage of self-sown germinants to become established and survive than those sown by hand in quadrats. The pre-

priming and watering in of hand sown seed may have induced the germination of seed in unsuitable microsites that may not have germinated otherwise and therefore died before establishment (Battaglia and Reid 1993). This may have been the case with *E. amygdalina* seed as germination proceeded fairly soon after sowing at Epping Forest and Evercreech. However, more germination of self-sown seed occurred at the Oatlands sites in the first six months than in the manually sown plots. The delay in germination of sown seed at Oatlands until the next winter/spring after sowing may have been due to dormancy in the *E. pauciflora* seed induced by insufficient stratification and/or cold dry weather immediately following sowing (Boland *et al.* 1980; Beardsell and Mullett 1984).

Seasonal effects

Every site had some germinants in the sown plots and/or from self-sown seed which was seen for the first time at the one-year since sowing census, showing that the seed beds remained receptive and some seed remained viable for at least that long. The delay in the germination of *E. pauciflora* seed as well as a second flush of *E. amygdalina* germination in the winter/spring season the year after sowing shows the critical role seasonal climatic conditions play in seed dormancy, germination and establishment (Stoneman 1994; Battaglia 1996). All research locations were particularly dry in the 2008-2009 summer that followed sowing, while the second half of 2009 was much wetter than previous years which had below average annual rainfall (see Figures 6-1 and 6-2). Both the seasonal timing and year of sowing can therefore be very influential on seed germination and on the progression to establishment (Battaglia 1996). Indeed Curtis (1990) suggests that above average rainfall for the first three months post emergence is needed to allow successful eucalypt recruitment. If this experiment had been undertaken a year later, very

different results may have been obtained solely due to differences in rainfall following sowing (Vaughn and Young 2010).

5.4.3 Site effects

While the quantity of germination that was observed at the experimental sites was strongly correlated with the structural complexity of sites, the number of seedlings surviving one year after sowing was not. Site was still a significant factor in seedling survival at one and two years respectively but the ranking of the sites in the structural complexity index did not match the ranking of sites in respect to the number of survivors.

The most meaningful comparisons for seedling survival can be made between the two sites at one location. The relatively healthy Tom Gibson Reserve site had significantly more plots containing seedlings and greater numbers of hand sown germinants and seedling survivors than the adjacent degraded Valleyfield site by the end of one year since sowing. However the survival rate of germinants at Tom Gibson over one year (8.7%) was the same as the Valleyfield site (8.6%) over two years showing that, proportionally, the degraded site did not do any worse than the healthier site. This suggests that this degraded site lacked some microsite elements that helped promote germination at the healthy site but the factors influencing the transition from germination to establishment were similar in both sites.

This is in contrast to the Oatlands sites at the second year census. Bald Hill and Liliesleaf were of similar structural complexity, but Bald Hill had significantly more recruitment in a greater amount of plots and a higher survival rate of manually sown seedlings and self-sown seedlings. Poor recruitment at the end of the two years (with only four plots at each site containing seedlings) was more comparable at Liliesleaf

and Valleyfield which were more structurally and compositionally different from each other. The measured factors that most distinguished Bald Hill from the other degraded sites was the relative lack of exotic weed species and *Lomandra* in all treatment plots compared to unfenced control plots within the remnant and the presence of seedlings of the native tall shrub/tree *Acacia dealbata* in some of the plots. This points to the importance of competitive (Standish *et al.* 2008; Skinner *et al.* 2010) and possibly early facilitative effects (Gomez-Aparicio *et al.* 2005) of other species.

Within each site, treatment effects were very variable among plots with occasional plots having a number of surviving seedlings, others only having one or two and many having none. This patchiness is not unlike the natural situation in dry woodland and forest where not all potential recruitment microsites are occupied (Schupp 1995) and recruitment is spatially restricted (Duncan 1999; Ashton 2000).

In combination, these results suggest that at the plot and microsite scales, variation in biotic, edaphic and microclimatic conditions not measured in the broad scale structural complexity index play a critically important role in eucalypt recruitment from seed.

5.4.4 Assessing recruitment success

Less than 1% of estimated viable seed sown resulted in an established seedling at all sites at one year after sowing, except Liliesleaf (1.33%) with most individuals recorded as a recent cohort. These figures are similar to survival of three eucalypt species after five years in a seedbed experiment in grassy woodlands in the New England Tablelands of NSW, as reported by Clarke (2002). All three degraded sites assessed in the current study were below 0.6% survival at the second year census.

This is considerably less than the likely 5% establishment from broadcast seed in logged dry forests suggested by forestry manuals (Lockett 1991) but considerably more than in two separate seedbed trials at comparable Midlands sites (Pinkard 1992; Fensham and Kirkpatrick 1992). These studies had slightly lower sowing rates and much greater sowing rates respectively than in the current study and each resulted in no to very limited eucalypt recruitment. Close and Davidson (2003) reported on three relatively successful mixed species direct sowing revegetation efforts in the Midlands, the success of which they attributed to good rain prior to or just after sowing, as well as effective post-planting weed control and watering. Subsequently however, the eucalypts were either preferentially browsed or were outcompeted by other species.

Relative to other work in the Midlands therefore, the current sowing trials were successful in the short term. Eucalypt seedlings were established from hand sown and self-sown seed in at least half or more of the treatment plots at all but one site one year after sowing, and in at least a quarter of the plots at three degraded sites after nearly two years. However, the total number of individuals established at some sites was small (5 at Valleyfield and 7 at Liliesleaf after two years).

In Chapter 2, fifty regenerating stems per hectare was identified as the minimum number required in a woodland stand and Gibbons *et al.* (2008b) suggested a minimum of 2 recruits per adult as the minimum number needed to maintain populations of scattered trees in agricultural landscapes in the long term. In the current study the 16 treatment plots equated to 0.04 ha within a 1 ha area and therefore a minimum of 3 established seedlings per plot would be required to reach the minimum 50 regenerating stems per hectare at sites that had no existing regeneration. Combining the amounts of hand sown and self-sown seedlings which

were observed alive at each site at the first year census, every site except Valleyfield had more than 50 seedlings. However this data were influenced by the germination of a new cohort just prior to the census. The second year results at three of the sites suggests that at least some, and quite likely a large proportion of, these new recruits would have died over the following summer.

By the end of the second year after sowing, of the three degraded sites assessed only Bald Hill, with 37 established seedlings, had anywhere near enough recruitment. With the number of adults per hectare at Bald Hill being approximately 57 and with the site containing approximately 3 existing regenerating stems per hectare (see Table 5-2), the resulting ratio of regeneration per adult was only 0.7:1 i.e. less than population replacement levels. This is, however a marked improvement on the ratio of 0.06:1 at this site prior to treatment (see Table 5-2).

In contrast, recruitment at Liliesleaf and Valleyfield after two years was only about a tenth of that required to restore regeneration to population replacement levels. This suggests that the techniques used in this study did not sufficiently identify or break down all of the barriers to recruitment at these sites and that further research into restoration techniques is required to improve eucalypt regeneration by seed at a range of degraded sites. However a number of lessons have been learnt and new hypotheses developed from these small scale seed sowing trials that may be used to direct larger scale adaptive management experiments (Cummings *et al.* 2007; Suding and Hobbs 2009b). These are discussed further in the synthesis contained in Chapter 7.

5.4.5 Conclusions

Cultivation and burning treatments each provided a seedbed for hand sown seed and natural seed rain that was superior to untreated ground, implying that some of the barriers to eucalypt recruitment had been removed, at least to some extent (Suding *et al.* 2004) and that some links in the recruitment chain had been restored (Young *et al.* 2005). The equal success of establishment techniques burning and cultivation respectively, with or without the addition of coarse woody debris, suggests that the recruitment niche of woodland eucalypts can be artificially expanded (Young *et al.* 2005) beyond that experienced in essentially natural situations (as described in Chapter 3). However each technique showed positive and negative aspects that may require further research to clarify particular processes at play. The generally low quantities and patchiness of recruitment experienced at experimental sites has consequences for the amount of restoration effort needed for only small gains.

Ashton (2000) declares that eucalypt regeneration in dry forest and woodland settings relies on complex but favourable chance events (i.e. the strong stochastic element). Results from this study have confirmed that establishment eucalypt regeneration from sown seed is influenced by a multitude of factors and that artificially recreating the recruitment niche is not an easy task, either in healthy or degraded remnants, with quite different outcomes to be experienced at different sites and at different times. The reliance of eucalypt establishment from seed on the juxtaposition of so many large and small scale factors suggests that bypassing the germination and early establishment niches by using planted seedlings as a restoration technique may provide better outcomes. The results of the second part of this study using planted seedlings are presented and discussed in the next chapter.

Chapter 6 Restoring the eucalypt persistence niche

6.1 Introduction

Planting seedlings bypasses the reproductive, dispersal and germination niches (Young *et al.* 2005) and may be the best option to restore eucalypt regeneration where these niches are restricted. However, the success of tree plantings in dry agricultural landscapes with variable rainfall is often below expectations (Yates *et al.* 2000a; Close and Davidson 2003; Close *et al.* 2010b). The degrading processes and abiotic and biotic factors identified in the previous chapter as being potential barriers to eucalypt recruitment from seed in these regions are also likely to be potential barriers to establishing regeneration through the planting of seedlings (Close and Davidson 2003). An additional risk is that of "transplant shock" which can occur when containerised seedlings are planted into new environmental conditions (Burdett 1990; Close *et al.* 2005a).

Within-plant characteristics (such as root: shoot ratio, leaf morphology, nutrient and carbohydrate reserves), abiotic stress factors (such as drought and frost), and interaction with the biotic and abiotic elements of the surrounding environment, all influence whether a newly planted seedling survives the shock of transplanting (Close *et al.* 2005a). Water stress in particular can limit early growth or cause mortality (Burdett 1990; Yates *et al.* 2000a; Grossnickle 2005; Close *et al.* 2009b). Newly planted seedlings have root systems which occupy a volume of soil that is 10-fold less than that of a seedling of similar top growth that has established from seed *in situ* (Burdett 1990). Access to soil water and new root growth during the first season in the field following planting are therefore extremely important for planted seedling survival. Seedlings can only enter the persistence phase when they are fully

'coupled' into the site hydrological cycle whereby water flows from the soil to plant roots, through the plant and into the atmosphere (Grossnickle 2005).

Abiotic and biotic variables that affect water availability at a planting site (including soil water repellency) are critical in the acclimatisation of planted seedlings to new environmental conditions (Close *et al.* 2005a). However, as discussed in previous chapters many remnant woodland sites have undergone major structural, compositional and biophysical changes that have negative consequences for water availability and eucalypt regeneration (Yates *et al.* 2000b; Close and Davidson 2004; Close *et al.* 2008). These changes necessitate the use of restoration techniques that aim to capture resources and increase their retention, particularly moisture (Yates *et al.* 2000a; Close and Davidson 2003). It is of interest to ascertain whether planted seedlings respond to these restoration techniques in a similar way at sites differing in forest condition or if, as suggested by Suding *et al.* (2004), the trajectory of restoration in degraded sites differs from that in less modified states.

Soil amelioration and weed control were shown in the previous chapter to be essential to overcome competition for resources and support eucalypt recruitment from seed within degraded remnants. This has also been shown to be necessary for successfully establishing planted seedlings in plantations on ex pasture sites in the Tasmanian Midlands (Close and Davidson 2003; Close *et al.* 2005b; Close *et al.* 2010b). Best-practice plantation techniques aim to increase moisture retention through the reduction of soil compaction and competition from grasses and weeds (Close *et al.* 2005b; Davidson and Close 2006). Davidson and Close (2006) and Close *et al.* (2010b) suggested best-practice for establishing environmental plantings in dry areas of Tasmania as being: application of knockdown herbicide; cultivation of the planting site by ripping and mounding ; a fallow period of 12 months (to allow

for soil moisture accumulation) prior to planting; application of knockdown and residual herbicides prior to planting in spring; fencing to control stock and browsing animals; planting of local genotypes of hardy seedlings of good quality; correct planting technique; watering seedlings in (where water is available); fertilising only if in unimproved pasture; and post-planting weed control. In the current study these techniques were adapted for use on a smaller scale with ripping and mounding being replaced by cultivation with a rotary hoe to avoid damage to standing trees (as described in Chapter 5). Due to time constraints the plots were only left fallow for a few months and herbicide was only applied once prior to cultivation and once post-planting. The use of fertiliser was deemed unnecessary as the degraded sites used in this experiment were likely to be nutrient enriched through a history of fertilisation and livestock grazing (Close *et al.* 2008).

The use of intense fire as an establishment technique was shown to have both positive and negative consequences for eucalypt recruitment from seed in the previous chapter. Other studies on planted seedlings in grassland and ex pasture sites have shown mixed effects of fire on the growth and survival of planted seedlings (Li *et al.* 2003; Close *et al.* 2005b). Ruthroff *et al.* (2010) reported improved weed control and establishment and growth of *Eucalyptus gomphocephala* seedlings planted in ashbeds compared with those in ripped and control plots in a restoration trial in degraded Western Australian Tuart woodlands. This suggests that the benefits afforded by ashbeds to naturally recruited seedlings (Chambers and Attiwill 1994) may also be of benefit to planted seedlings. Rapid growth of seedlings may give them an advantage over competitors and allow them to recruit to the canopy (Forestry Commission 1993; Florence 1996).

The presence of coarse woody debris was previously shown to be an important element in the regeneration niche of Tasmanian woodland eucalypts but not to be so ubiquitous in the microsites containing lignotuberous spouts (Chapter 3). While forestry manuals suggest leaving the unburnt crowns of trees in logged forests to protect new seedlings (Orr 1991; Orr and Todd 1992), it does not appear that the effect of the presence of coarse woody debris on planted seedling survival has been tested in a restoration context prior to this study.

In a trial of the success of planted seedlings of various local and native species into ex pasture sites in the Tasmanian Midlands, Close et al (2010a) found that different species had different sensitivities to environmental variation and had different capacities to gather resources and occupy a site. This suggested that more than a single species should be used in subsequent trials. Therefore two local co-occurring species, one in the subgenus *Symphyomyrtus* and the other in *Eucalyptus* (previously called *Monocalyptus*), were used in the experiment reported here in order to investigate potential species differences in the effects of establishment treatments in sites of varying condition.

The process of woodland eucalypt regeneration in Tasmania, as described in Chapter 3, includes an ontogenetic niche shift when the restricted seedling recruitment niche broadens over time to become the juvenile persistence niche within which lignotuberous sprouts persist in conditions similar to the general forest floor. The overall aim of this chapter is to assess whether seedlings planted in woodland remnants establish and perform best in conditions mimicking those where eucalypts naturally recruit or where plantation style establishment techniques have been used.

The specific research questions of this study were:

1. How does the condition of a site influence the early survival of planted eucalypt seedlings?

2. Are there differences in early survival and growth of seedlings planted in different establishment treatments that mimic elements of the natural recruitment niche (ashbeds and coarse woody debris) or utilise plantation establishment techniques (cultivation/herbicide)?

3. Do establishment treatments affect the survival and growth of two different naturally co-occurring species in the same way?

4. Does soil water repellency affect survival of planted seedlings?

5. How do structural elements of remnants effect seedling survival?

6.2 Methods

6.2.1 Experimental design

Experimental sites, establishment treatments and planting dates were the same as for the seed sowing experiment described in Chapter 5. Descriptions of the sites are given in Table 5-1and mean and standard error values for structural complexity attributes (measured for the study described in Chapter 2) are given in Table 5-2. There were a total of six sites: two healthy sites (Tom Gibson Reserve at Epping Forest and Evercreech coupe at Fingal) and four degraded intermediate sites (Valleyfield at Epping Forest, Evercreech plantation remnant at Fingal, and Bald Hill and Liliesleaf at Oatlands), as categorised in Chapter 2.

Four replicates of four establishment treatments at each site were planted: burnt with logs, burnt no logs, cultivated with logs, cultivated no logs (described in section 5.2.2 and pictured in Figure 5-2). Each plot was fenced and 5m x 5m in size. A total of 352

seedlings were planted at each site, 172 each of *Eucalyptus amygdalina* and *Eucalyptus viminalis*.

Treatments with added logs had, eight seedlings (four *E. amygdalina* and four *E. viminalis* in random positions) planted 30cm apart on the northern and southern sides of each log (Figure 6-1). This gave a total of thirty two seedlings in each plot with logs (sixteen of each species). Seedlings were planted 15cm away from the log because this was the average distance from coarse woody debris that seedlings were found in the first two sites from the seedling microsite study reported in Chapter 3.

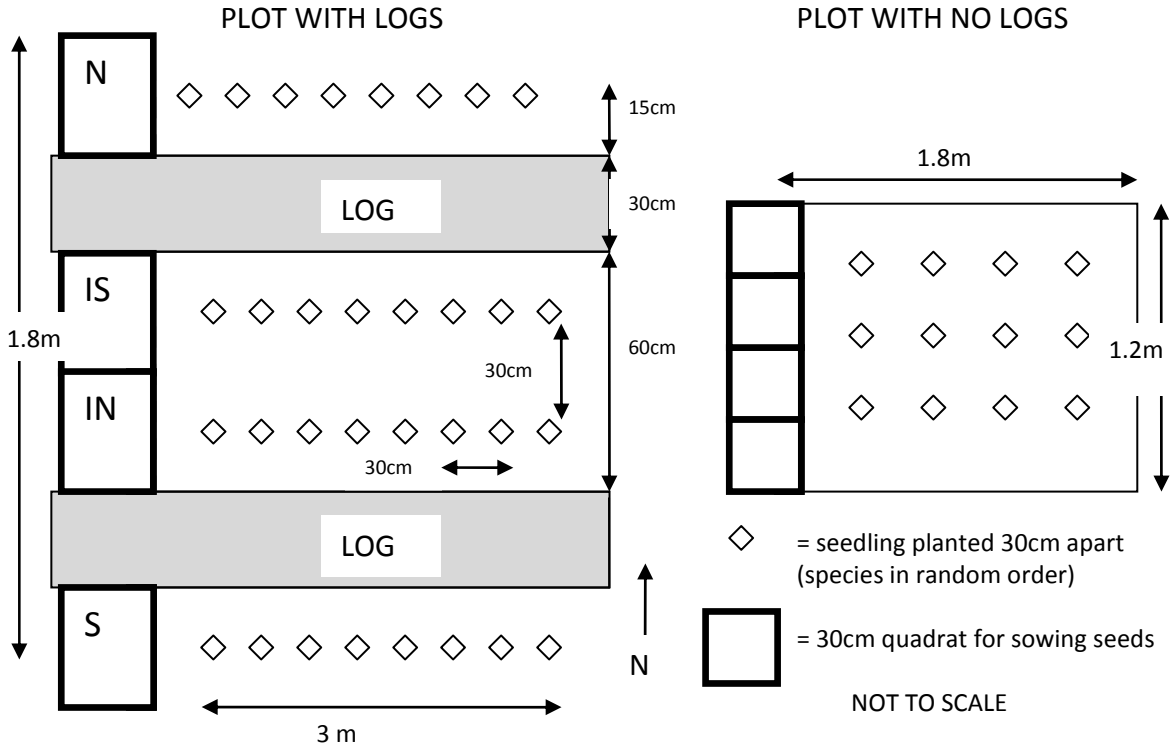

Figure 6-1 Schematic diagram of plot layouts (not to scale) of treatment plots with logs and without logs showing positions of planted eucalypt seedlings and quadrats sown with locally collected eucalypt seed. Letters inside quadrats indicate microsite aspect relative to logs (N= north, IS= inner south, IN=inner north, S=south).

Plots without logs each had twelve eucalypt seedlings (six of each species in random positions) planted within a 30 x 30cm east/west aligned grid in the centre of the plot (Figure 6-1). Fewer seedlings were planted in non log plots due to restriction in the

total number of seedlings available. Seed of both *E. viminalis* and *E. amygdalina* were collected from healthy trees in the Evercreech Valley. Seedlings of the two eucalypt species were grown under identical conditions at Woodlea nursery in northern Tasmania. Seedlings were grown in 40 cell (40mm wide by 90mm deep) plastic trays (Hiko) in a standard forestry potting mix. Seedlings were hardened off in an outdoor nursery area in by with withholding nutrients and providing adequate but limited water (pers. comm. Julie Target). The seedlings were approximately 18-20 months old and ranged in height from 13.2 to 63.9 cm with an average at planting of 38.5 ± 0.2 cm for *E. viminalis* and 34.9± 0.2 cm *E. amygdalina*.

Each seedling was planted by hand using a trowel such that the potting mix of the root ball was well covered by soil and root ball-soil contact was adequate (Close *et al.* 2005a). The seedlings were then watered separately using a watering can. Watering seedlings in at planting was regarded as an essential component of best practice revegetation by Close and Davidson (2003) in their review of improved plant establishment in the Midlands. The seedlings were not watered again for the duration of the experiment.

Seedling height (cm) and health were recorded at planting (September/October 2008) then seedling height was measured every 3 months and health/survival assessed monthly for six months (up to March 2009), again at 9 months after planting (June 2009) and finally 12-13 months after planting (September- November 2009).

The three most degraded sites (as measured by structural complexity), Bald Hill and Liliesleaf at Oatlands and Valleyfield at Epping Forest, were assessed again for seedling survival approximately 22 months after planting.

Measurements of distance from each planted seedling to nearest live tree were made at each site except Evercreech coupe. Methods for determining soil water repellency and the resultant median water repellency (WDPT) class of each site are the same as those reported in the previous chapter (sections 5.2.7 and 5.5.7 respectively).

Seedling health was assessed with a subjective visual ranking of 0-5 based on foliage density, colour, and level of foliar herbivory and/or fungal attack with 5 being most healthy and 0 being dead.

6.2.2 Data Analysis

Daily rainfall data for the three years prior to planting (2006-2008) and two years after planting (2009-2010) were obtained from the Bureau of Meteorology climate online website (http://www.bom.gov.au/climate/data/) for the closest weather stations to each of the research sites (Forton at Epping Forest, Fingal High Street, Oatlands Post Office). The total annual rainfall for each year and the percentage of long term annual average rainfall were determined. Maximum and minimum temperature data were also obtained.

Spearman's rank correlation tests were undertaken to determine whether survival percentage of planted seedlings one year after planting at the six research sites was associated with structural complexity index scores or individual structural attributes of the sites (listed in Table 5-2).

Survival analysis of the planted eucalypt seedlings was undertaken as these analyses can be used to compare groups over the entire distribution of times to event (in this case seedling death) rather than comparing averages at fixed points in time (Fox 2001). Survival analysis takes into account data that is censored (i.e. the exact time

of death is unknown and individuals may survive past the end of the study) and event/ death times that are not normally distributed under any standard transformation (Muenchow 1986; Fox 2001).

The *hazard function h(t)*, or the conditional mortality rate, is the conditional probability that a death occurs at exactly time *t* given that it has not occurred before then, that is, $P(T=t \mid T \geq t)$ (Muenchow 1986). The *survival function* $S(t)$ is the probability that an individual survives beyond age *t* ,that is, $Pr(T>t)$ where T is a continuous random variable representing the age of death (Beckage and Clark 2003). The survival function is also known as the cumulative survival rate and the graph of $S(t)$ is the survival, or survivorship, curve (Lee and Wang 2003).

The Kaplan-Meier product-limit method (Kaplan and Meier 1958) for estimating the survivorship function is the most commonly used method for estimating survival functions in ecology (Muenchow 1986; Lee and Wang 2003). It is the default method for fitting survival curves for censored data in R using the function `survfit` (Therneau and Lumley 2008) and was used in this analysis. This function produces a stepped survivorship curve that adds information as each death occurs (Crawley 2005). The survival curves are presented graphically with time since planting on the x axis rather than calendar dates. The time to death was recorded as the last day an individual was seen alive as the exact date of death was unknown (as suggested by Harrington and Fleming 1982).

The survival curves of two or more samples can be compared using nonparametric tests that are based on the rank ordering of survival times. In this analysis the Peto and Peto's generalized Wilcoxon test was used. This test is sensitive to early rather than late deaths and has the most power (compared to alternative tests such as the log

rank test) when the hazard functions are not parallel (Fox 2001), as was the case in this study. Post hoc pair wise comparisons among groups were made using the same test to ascertain which groups were significantly different from each other. These analyses were run in R using the `survdiff` function with ρ=1 (Therneau and Lumley 2008) to determine whether there were differences in the survival of planted eucalypt seedlings over time among location, among sites, between species and among treatments.

Cox proportional hazard regression analysis (`coxph` function in R) was used to separately model the influence on the hazard of a seedling dying of the continuous variables initial height of seedlings, distance from tree (where it was measured) and water repellency (as measured by WDPT, Chapter 5). This regression analysis could not to be used to model the influence of site, species or treatment on seedling survival over time (and thus could not develop a multivariate overall model) as the hazard functions of these predictors were not parallel over time. Proportionality of hazards is an essential assumption of the Cox regression analysis (Fox 2001) and was tested for each predictor using the `cox.zph` test in the Survival package in R.

Relative growth rate ([final height- initial height]/ initial height *100) and health of seedlings still alive at the last census date data were analysed by ANOVA with species, site and treatment as fixed factors. Relative growth rate was square root transformed to improve normality of distribution. Correlations between mean relative growth of surviving seedlings and the soil water repellency (WDPT) in each plot and between mean seedling health and WDPT were assessed using Pearson's product-moment correlation tests.

6.3 Results

6.3.1 Weather

The three years up to and including the year of planting (2006-2008), were all below long term average annual rainfall at all of the six research sites (Table 6-1). The month after planting, October 2008, was also particularly dry with only 16.4mm of rain falling at Oatlands, 14mm at Epping Forest and 8.2mm at Fingal. The year after planting (2009) was a relatively wet year with the majority of the rain falling in the last 6 months. However, very little rain fell at all locations in January 2009 (8.1mm at Epping Forest, 5.4mm at Fingal and 3.6mm at Oatlands) with February and early March continuing to be dry, particularly at Epping Forest.

Table 6-1: Total annual rainfall (mm) and percentage of the long term average rainfall for the years 2006 -2010 at the three locations of the six research sites (closest meteorological station number in brackets): Epping Forest (Tom Gibson Reserve, Valleyfield), Fingal (Evercreech coupe and plantation) and Oatlands (Liliesleaf and Bald Hill).

Location and long term average annual rainfall	Year	total rainfall (mm)	% of long term average
Epping Forest	2006	416	73.8
563mm	2007	412	73.1
(091032)	2008	466.6	82.8
	2009	586.9	104.1
	2010	614.3	109.0
Fingal	2006	352.8	58.1
609mm	2007	468.8	77.2
(092012)	2008	476.4	78.5
	2009	727.8	119.9
	2010	698.8	115.1
Oatlands	2006	317.9	57.7
550mm	2007	380	69.0
(093014)	2008	425.6	77.2
	2009	696.6	126.4
	2010	498	90.4

There were numerous frosts at all three locations immediately following plantings which continued until January 2009 (Figure 6-3). The hottest day on record (40.6°C) was experienced at Fingal and Epping Forest in late January 2009 (BOM 2011).

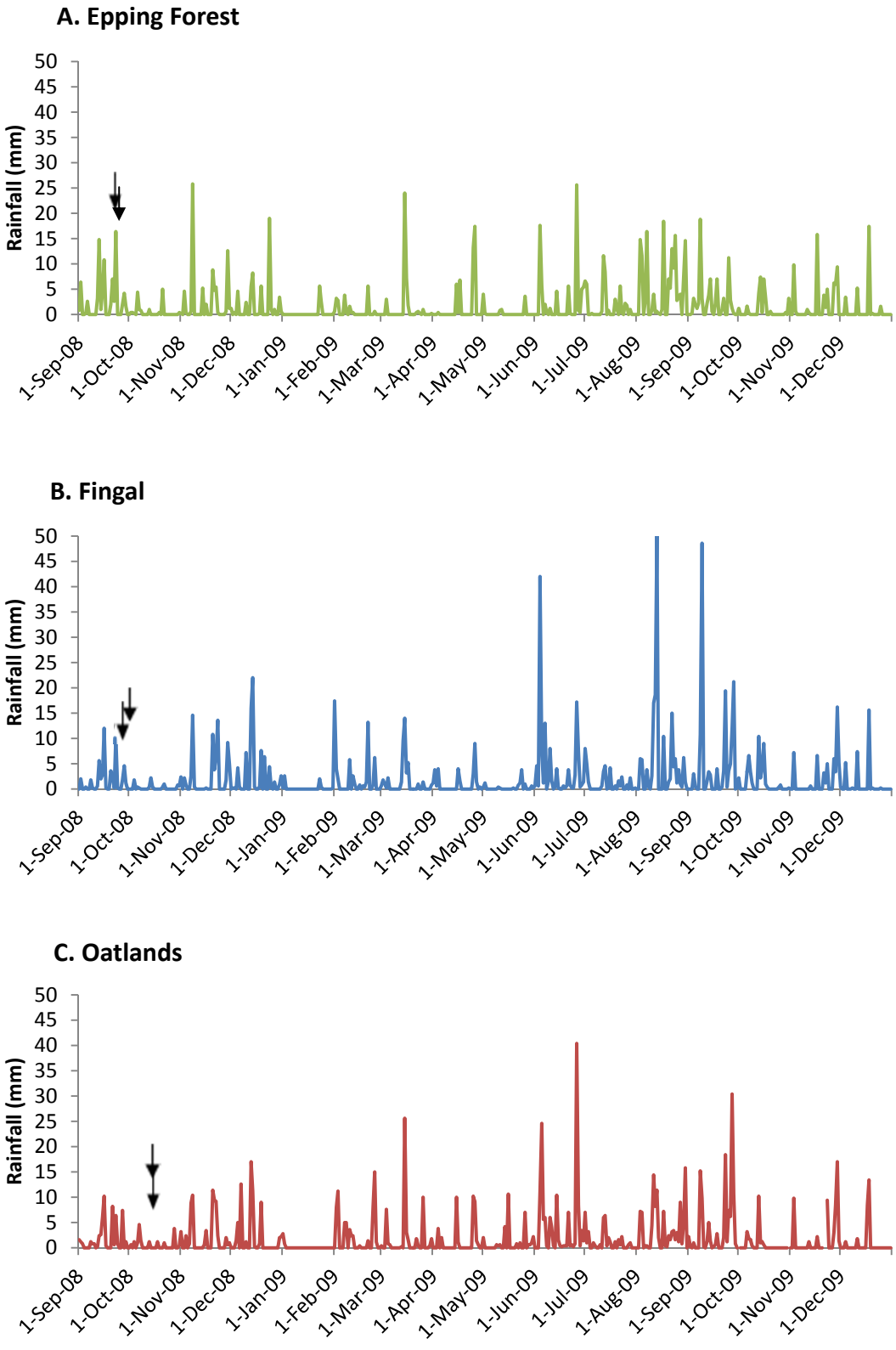

Figure 6-2 Daily rainfall (mm) over the duration of the first year following planting, recorded at the closest meteorological station to the six research sites (see table 5.3 for station numbers): A) Epping Forest (Tom Gibson Reserve, Valleyfield), B) Fingal (Evercreech coupe and plantation) and C) Oatlands (Liliesleaf and Bald Hill). Arrows indicate planting dates.

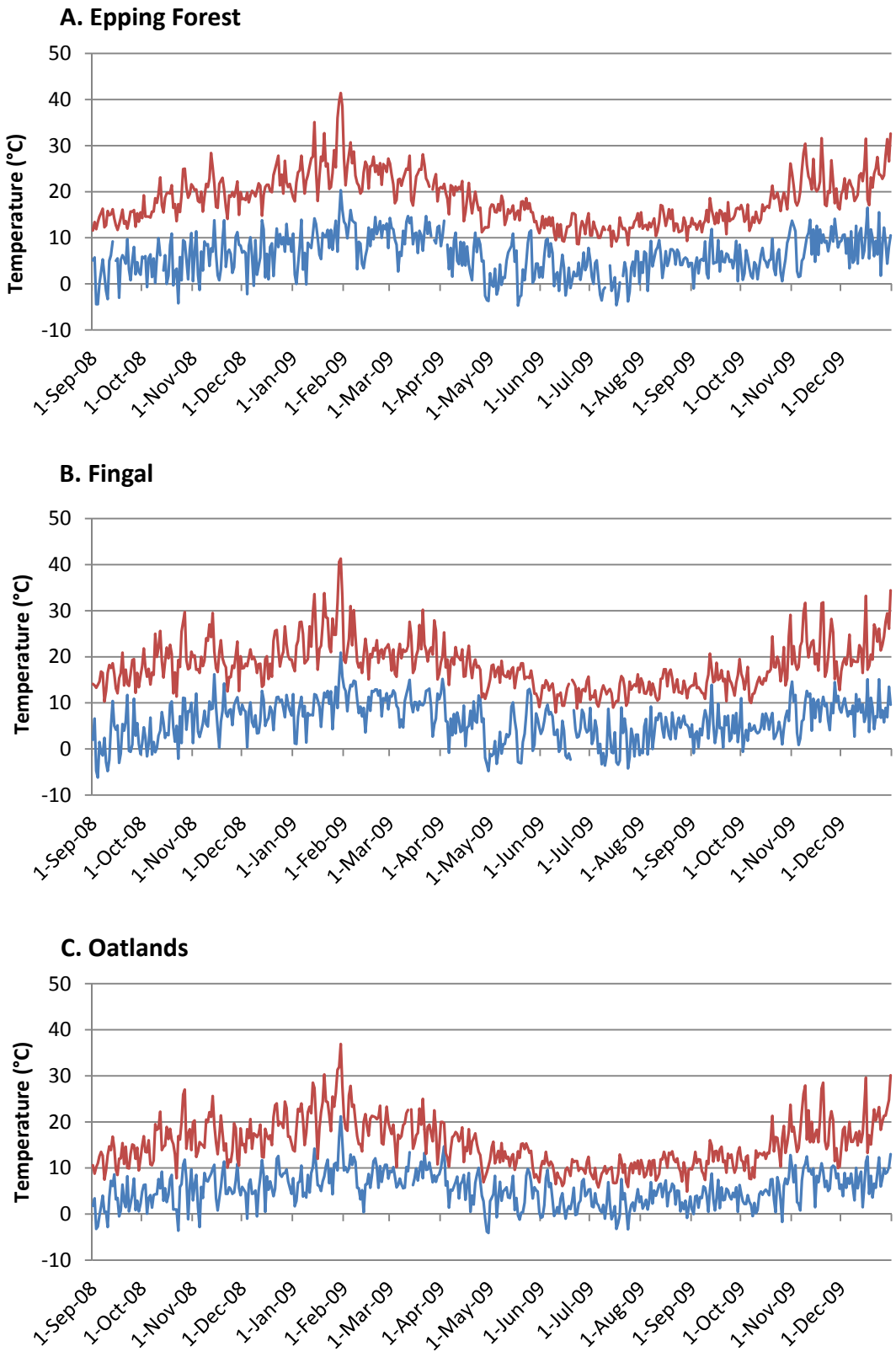

Figure 6-3: Maximum (red lines) and minimum (blue lines) daily temperatures (°C) over the duration of the first year since planting at the closest meteorological station to the six research sites (see table 5.3 for station numbers): A) Epping Forest (Tom Gibson Reserve, Valleyfield), B) Fingal (Evercreech coupe and plantation) and C) Oatlands (Liliesleaf and Bald Hill).

6.3.2 Structural complexity and seedling survival

Structural complexity index score of planting sites and percentage survival of planted seedlings a year after planting were strongly correlated (r= 0.868, p= 0.025). Seedling survival was positively associated with basal area (r= 0.897, p= 0.016) and number of large trees ha^{-1} (r= 0.888, p= 0.018) at a site. All other structural variables measured for the structural complexity index were not significantly correlated with planted seedling survival.

The percentage survival of planted seedlings at the end of the first year at each site was; Evercreech coup 93.2%, Tom Gibson Reserve Epping Forest 66.5%, Evercreech plantation remnant 53.4%, Liliesleaf Oatlands 50.3%, Valleyfield Epping Forest 31% and Bald Hill Oatlands 26.4%.

6.3.3 Soil water repellency and seedling survival

Median water repellency persistence (WDPT) categories of individual sites are shown in the previous chapter (Table 5-8). With data from all sites combined there was a significant ($r = -0.430$, $p<0.001$) negative correlation between plot level soil water repellency persistence and the percentage of planted seedlings surviving in each plot at the year census. This result was confirmed by Cox proportional hazard regression analysis which showed a significant relationship between seedling survival over time and potential soil water repellency ($W_5 = 97.3$, $p< 0.001$). For every increase in WDPT class there was a 20% increase in the hazard of a planted seedling dying.

There was a significant ($r = -0.303$, $p = 0.002$) negative correlation between soil water repellency and relative growth of surviving seedlings but not between water repellency and seedling health ($r = -0.177$, $p = 0.083$).

6.3.4 Survival analysis of planted seedlings

The survival curves for eucalypt seedlings which spanned approximately a year from planting were significantly different among locations (Fingal>Epping Forest>Oatlands χ^2 =245, df 2, p=0), research sites (χ^2 = 475, df 5, p=0, Figure 5.6), establishment treatments (χ^2 = 22.4, df 3, p<0.001) and between species (*E.viminalis*> *E.amygdalina* χ^2 =112, df 1 p=0).

Site

Survival curves of seedlings at Evercreech plantation remnant and Liliesleaf were not significantly different from each other, but all other combinations of sites were (Figure 6-4).

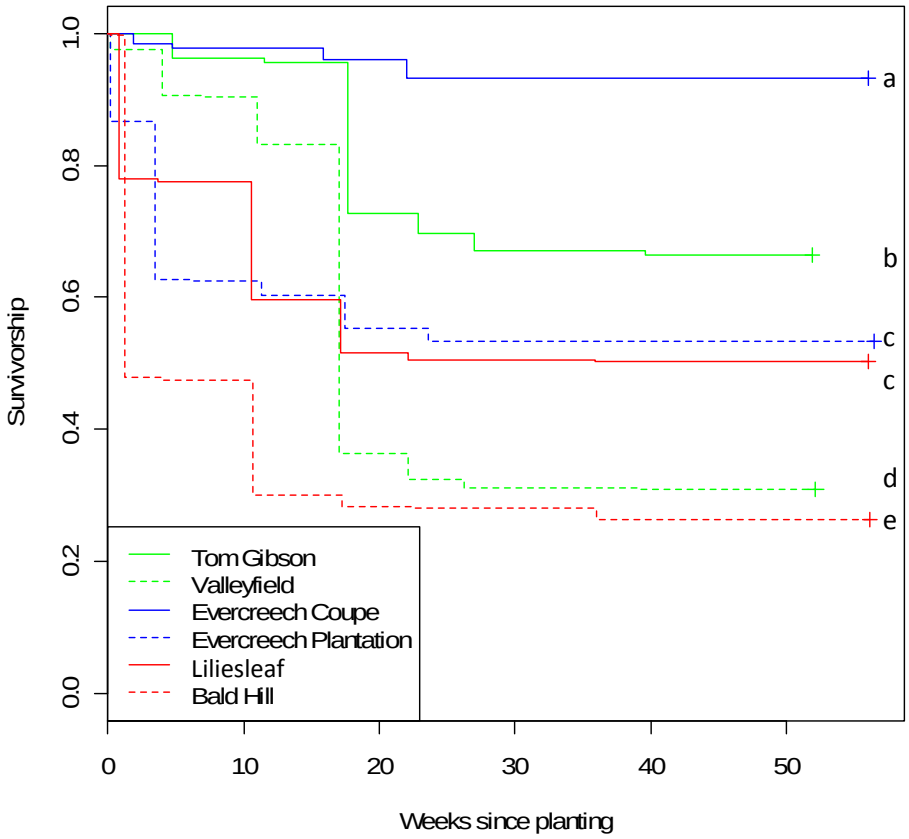

Figure 6-4 Survival curves of eucalypt seedlings planted in September/October 2008 at six research sites in the Midlands of Tasmania. The locations of sites are coloured green for Epping Forest, blue for Fingal and red for Oatlands. Significant differences among survival curves of seedlings at each site are indicated by different lower case letters

The patterns of times until death of seedlings were distinctly different between Valleyfield and Bald Hill despite them ending up with similar survival rates a year after planting. The survival curves of the seedlings from these two sites were therefore significantly different (Figure 6-4).

Almost half the seedlings planted at Bald Hill, and just under a quarter at Liliesleaf, died within a month of planting (October - November 2008). Another drop in survival occurred at the Oatlands sites over the summer (January -February 2009, Figure 6-4). At the other degraded site, Evercreech plantation, a third of seedlings had died within the first six weeks (October - November 2008) after planting (Figure 6-4). Seedlings at the Epping Forest sites, the degraded Valleyfield in particular, experienced dramatic mortality during January- February 2009 (18-22 weeks after planting, Figure 6-4).

Establishment treatment

Using combined data from all sites and both species, the probability of a seedling in the 'Cultivated with no log' treatment surviving over a year was significantly greater than seedlings in all other treatments ($\chi^2 = 22.4$, df 3, $p<0.001$, Figure 6-5). Other treatments did not differ significantly.

The survival curves in Figure 6-5 show that the hazard of dying for seedlings in different treatments changed over time. The hazard of dying was highest for seedlings in burnt plots in the first few months after planting. However, the survival curves cross at around 18 weeks after planting (mid-summer) when it became more hazardous to be in a cultivated plot with a log than in any of the other plots. After summer the survival curves stabilised with relatively few deaths over winter and spring.

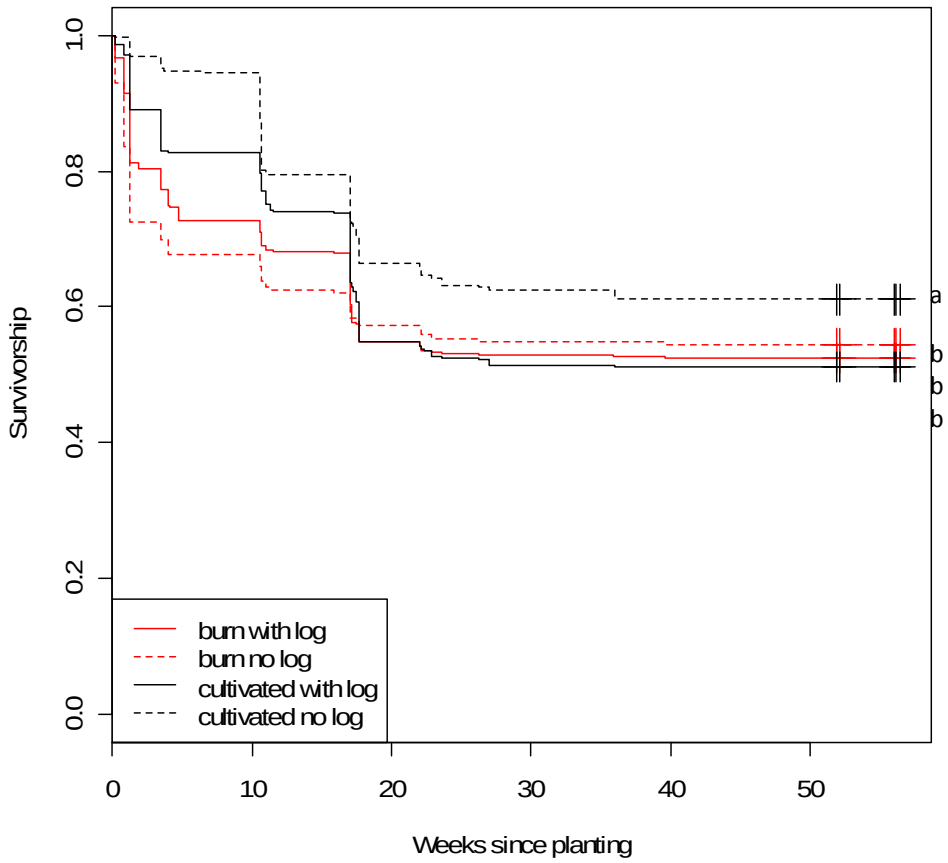

Figure 6-5 Survival curves of planted eucalypt seedlings (data from 6 sites combined) in four different establishment treatments. Significant differences among survival curves of seedlings in different treatments are indicated by different lower case letters.

Species

There was a significant difference in the survival curves of the two species across all sites with overall survival at year end being *E. viminalis* 65.8%, *E. amygdalina* 41%. There was also a significant difference between survival curves of the two species at every site individually (p values all <0.001) with the exception of Evercreech Coupe (p=0.085).

Accordingly the effect of establishment treatment on the survival of seedlings was also assessed on each species separately at each site (Table 6-2). The survival of *E. viminalis* seedlings was not significantly affected by establishment treatments at all

sites except Liliesleaf at Oatlands (Table 6-2). The survival of *E. amygdalina* was significantly affected by establishment at all sites except Valleyfield (Table 6-2).

Table 6-2 Percentage survival of *Eucalyptus viminalis* and *E.amygdalina* at the year census (treatments pooled) plus results of Peto and Peto's generalized Wilcoxon test on whether there were differences among the survival curves in the four different establishment treatments for each of the species at each of six research sites.

Site	Eucalyptus species	Total survival %	Treatment differences?	χ^2	p value
Tom Gibson Reserve	*E.viminalis*	87.0	no	7.1	0. 068
	E.amygdalina	46.0	yes	20.4	< 0.001
Valleyfield	*E.viminalis*	50.0	no	6.9	0.073
	E.amygdalina	11.9	no	7	0.076
Evercreech Coupe	*E.viminalis*	95.4	no	1.5	0.675
	E.amygdalina	90.9	yes	16.4	< 0.001
Evercreech Plantation	*E.viminalis*	63.1	no	5.1	0.162
	E.amygdalina	43.7	yes	15.8	0.001
Liliesleaf	*E.viminalis*	65.9	yes	18	< 0.001
	E.amygdalina	34.7	yes	32.9	< 0.001
Bald Hill	*E.viminalis*	34.7	no	8.8	0.052
	E.amygdalina	18.2	yes	20.4	< 0.001

At every site and for both species, with the exception of *E. amygdalina* at the healthy sites Tom Gibson Reserve and Evercreech coupe, seedling survival over the year after planting was highest in one of the cultivated treatments (Figure 6-6A and Figure 6-6B).

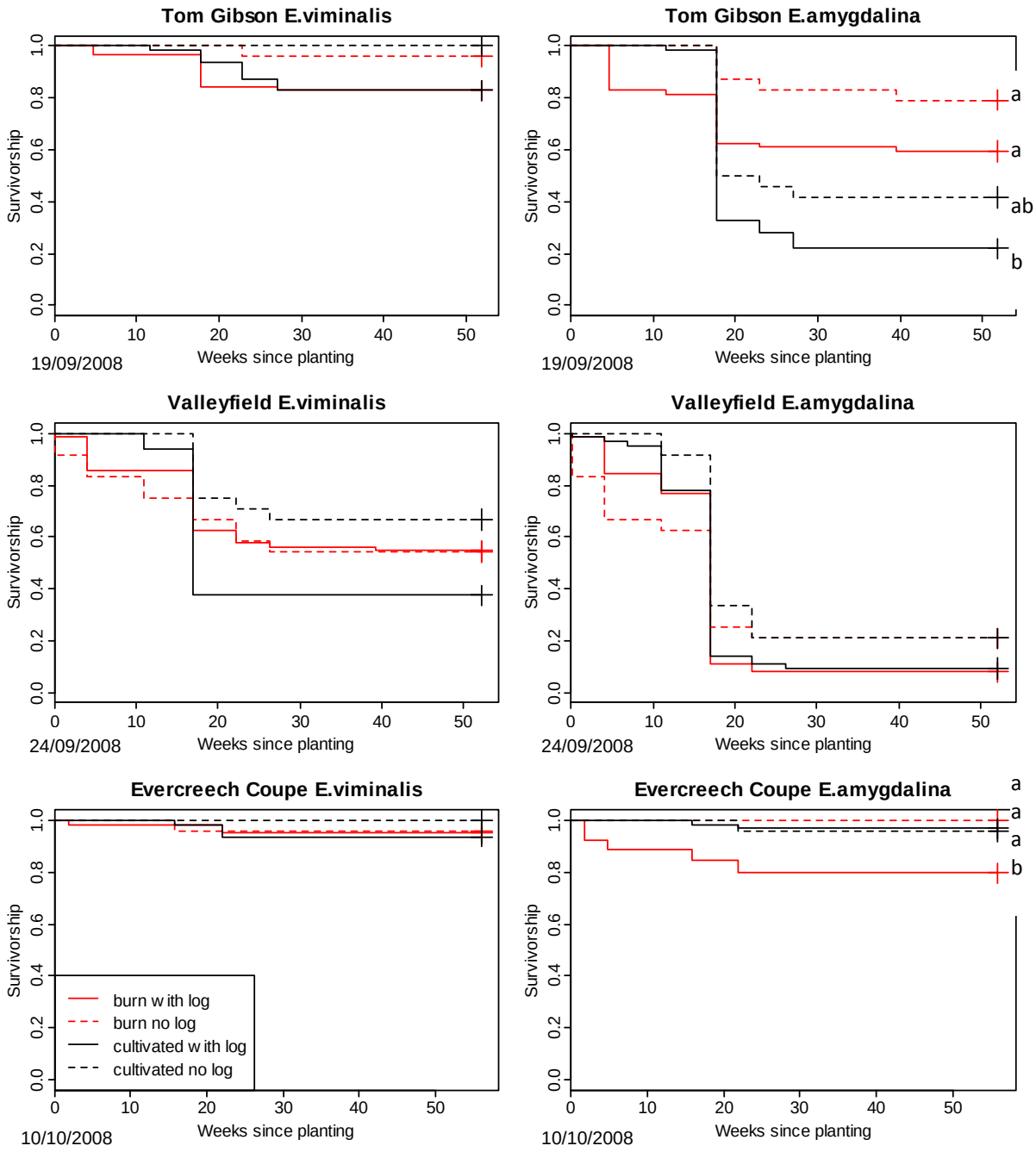

Figure 6-6A Survival curves for *Eucalyptus viminalis* and *E. amygdalina* in four establishment treatments at three sites: Tom Gibson Reserve and Valleyfield at Epping Forest and Evercreech Coupe at Fingal. The survival curves of *Eucalyptus viminalis* are not significantly affected by establishment treatments at any of these sites (p values Table 6-2). Significant differences among survival curves of *E. amygdalina* in different treatments at each site are indicated by different lower case letters. Planting dates are shown at time zero for each site.

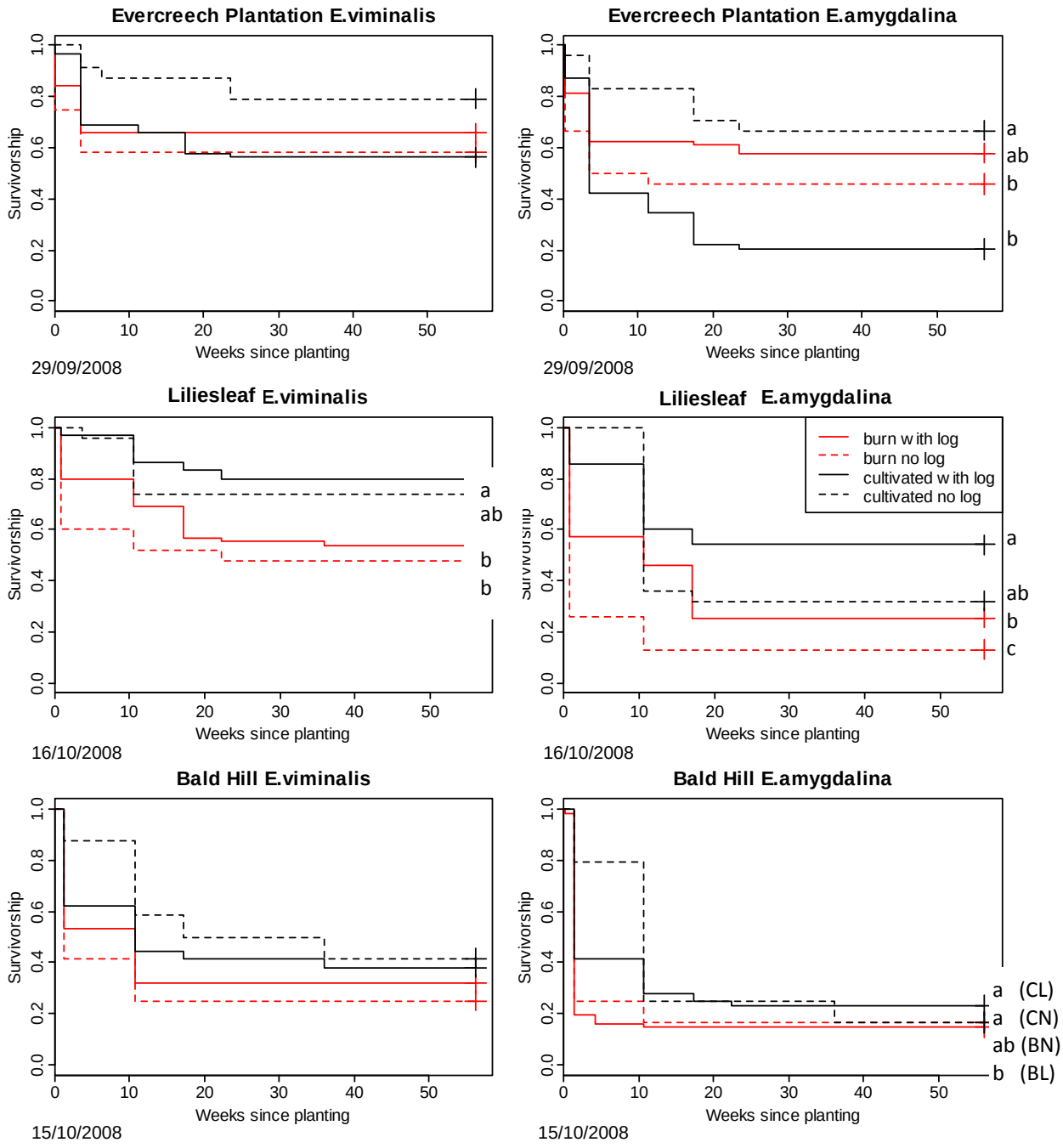

Figure 6-6B Survival curves for *Eucalyptus viminalis* and *Eucalyptus amygdalina* are significantly different at all three sites. The survival curves of *E. viminalis* are not significantly affected by establishment treatments at Evercreech Plantation and Bald Hill (p values Table 6-2). Significant differences among survival curves in different treatments at each site are indicated by different lower case letters. Planting dates are shown at time zero for each site.

2nd year survival

Between the censuses in November 2009 and September 2010 at Oatlands, there were five deaths of planted seedlings at Liliesleaf and two at Bald Hill with overall survival after two years being 48.9% and 26.4% respectively. There were no deaths of planted seedlings at Valleyfield between September 2009 and 2010 (31% survival overall). Treatment effects only changed between the first and second year census at Liliesleaf where seedling deaths in the burnt with log treatment changed it from being significantly different to survival in burnt without log plots at year 1 to not being significantly different in year 2.

Initial seedling height

There was no influence of the initial height of seedlings on seedling survival over one year ($\chi^2 = 0.02$, df 1, p= 0.878) based on Cox proportional hazard regression analysis.

Distance from closest tree

At all sites where distance from closest tree was measured there was a significant relationship between seedling survival and the distance to nearest tree, except for Bald Hill at Oatlands ($\chi^2= 0.22$, df 1, p=0.82). For every metre further away a seedling was from the closest adult tree the hazard of dying was reduced by: 11% at Tom Gibson ($\chi^2 = 4.71$, df 1, p= 0.03), 5.5% at Valleyfield ($\chi^2= 2.10$, df 1, p=0.035), 8.1% at Evercreech plantation ($\chi^2 =5.6$, df 1, p= 0.018), and 3.3% at Liliesleaf ($\chi^2=$ 3.22, df 1, p= 0.049). The closest a planted seedling was to a tree was 2m (average 6.6 ± 0.1m) at Tom Gibson, 6.5m (average 10.0 ± 0.1m) at Valleyfield, 5.2 m (average 9.4 ± 0.1m) at Evercreech plantation, 6.7m (average 12.7 ± 0.9m) at Liliesleaf and 8.3 (average 12.5 ± 0.9m) at Bald Hill.

6.3.5 Relative growth of planted seedlings

There was no significant difference in relative growth rate between the two eucalypt species (Table 6-3). The strongest effect on the relative growth of surviving seedlings a year after planting was treatment (Table 6-3). Seedlings in the burnt treatments had higher average growth rates than seedlings in cultivated plots. However, there were also significant site effects and a significant site by treatment interaction (Table 6-3).

Table 6-3 ANOVA of square root transformed seedling relative growth data from two species and six sites including fixed design effects and interactions

	Df	Mean Sq	F	p
Site	5	224.87	29.73	< 0.001
Species	1	3.47	0.46	0.49
Treatment	3	424.87	56.18	< 0.001
Site:Species	5	1.66	0.22	0.95
Site:Treatment	15	40.16	5.31	< 0.001
Species:Treatment	3	8.14	1.08	0.36
Site:Species:Treatment	15	9.52	1.26	0.22
Residuals	1021	7.56		

Due to the significant site by treatment interaction, the effect of treatments on relative growth of seedlings (species combined) was further explored by separate one-way ANOVA at each site. The relative growth of seedlings was significantly influenced by establishment treatment at all sites except Liliesleaf farm remnant at Oatlands (Table 6-4) where growth was relatively low across all treatments. Treatment effects varied among sites as shown in Figure 6-7.

Table 6-4 One way ANOVA results for differences among establishment treatments in the relative growth (% of initial height) of planted eucalypt seedlings at 6 research sites showing F values, degrees of freedom and p value for each analysis.

Site	Df	F	p value
Tom Gibson Reserve	3, 230	18.39	<0.001
Valleyfield Farm Remnant	3, 105	8.70	<0.001
Evercreech Coupe	3, 315	43.26	<0.001
Evercreech Plantation Remnant	3, 184	13.02	<0.001
Liliesleaf Farm Remnant	3, 158	0.782	0.505
Bald Hill Farm Remnant	3, 89	6.48	<0.001

At least one of the burnt treatments produced the greatest growth at all sites although this was not significant at the two Oatlands sites (Figure 6-7). Seedlings in cultivated treatments did not significantly differ in relative growth at all sites except Bald Hill.

Treatment effects were greatest at the two healthy sites (Tom Gibson and Evercreech coupe Table 6-4) which showed significantly greater seedling growth in the burnt treatments relative to both cultivated treatments (as did the more degraded Evercreech plantation remnant). The other sites showed mixed results (Figure 6-7).

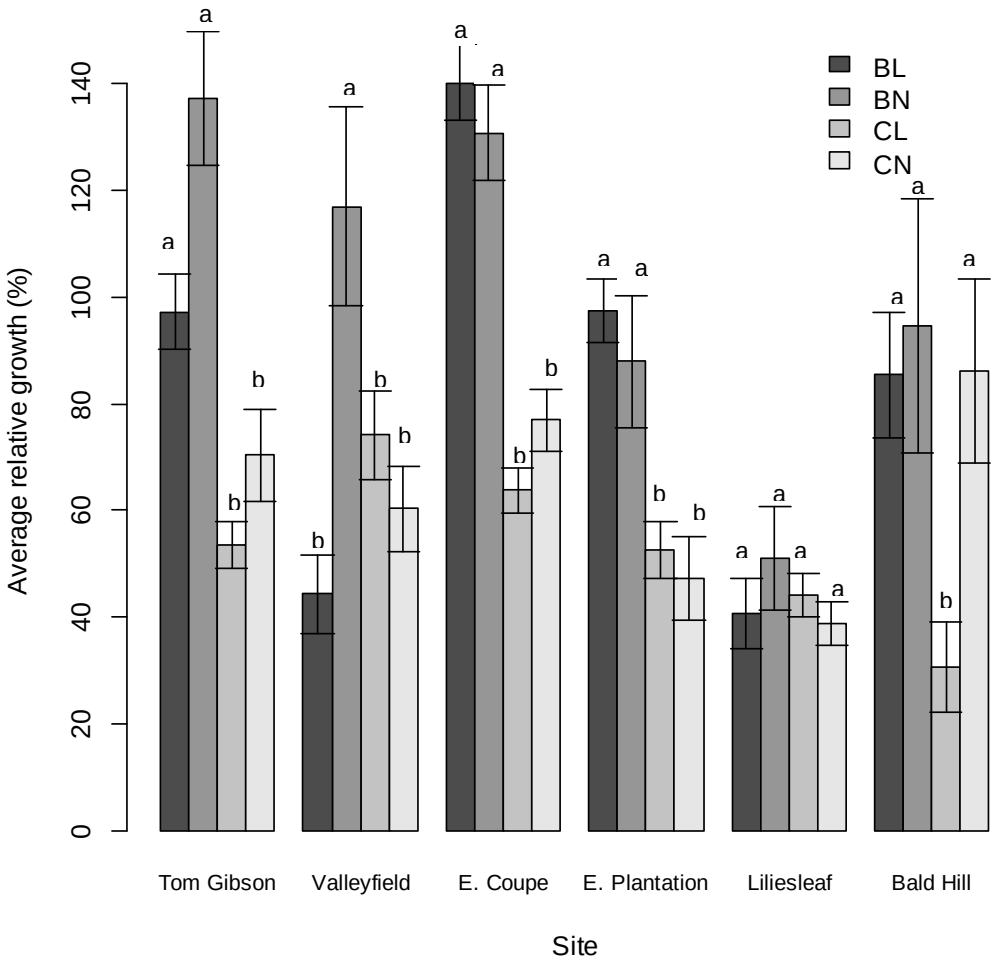

Figure 6-7 Relative growth (% of initial height) of eucalypt seedlings (species combined) surviving one year after planting at six sites in four establishment treatments on 5m x5m plots: BL = burnt with logs added, BN= burnt with no logs added, CL = cultivated with logs added, CN = cultivated with no logs added. Lower case letters indicate significantly different mean relative growth among treatments within each individual site.

6.3.6 Seedling health

There were complex effects of species, site and treatment on the health of surviving seedling with a significant site by species by treatment interaction (Table 6-5). The largest effect was the difference between species with surviving *E .amygdalina* averaging a health score of 3.4 ± 0.05 and *E. viminalis* averaging 2.9 ± 0.11 at the end of year. This was largely due to *E. viminalis* being more severely attacked by insects. Treatment effects varied among sites and species as shown in Figure 6-8.

Table 6-5 ANOVA of eucalypt seedling health including fixed design effects and significant interactions

Source	Df	Mean Sq	F	p
Species	1	56.09	56.73	< 0.001
Site	5	8.09	8.19	< 0.001
Treatment	3	46.88	47.41	< 0.001
Species:Site	5	5.06	5.12	< 0.001
Species:Treatment	3	3.48	3.52	0.015
Site:Treatment	15	4.76	4.82	< 0.001
Species:Site:Treatment	15	2.08	2.11	0.008
Residuals	1081	0.989		

Health of planted seedling was most affected by treatment at Tom Gibson reserve where both species were healthier in burnt than cultivated plots. Seedling health was not affected by treatment at Liliesleaf and was generally below average (Figure 6-8).

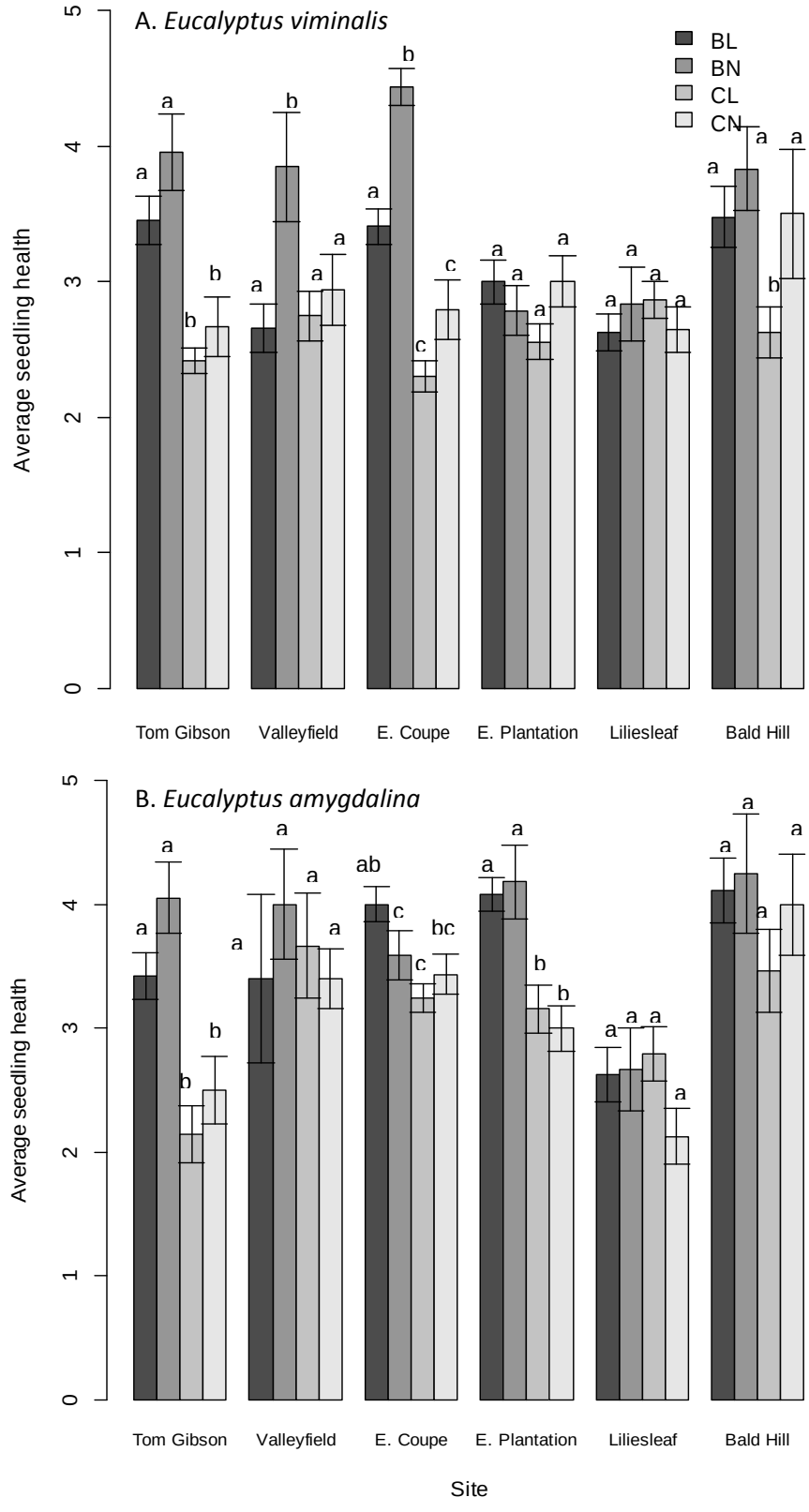

Figure 6-8 Average health of A. *Eucalyptus viminalis* and B. *Eucalyptus amygdalina* seedlings a year after planting at six sites in four treatments (BL = burnt with logs added, BN= burnt with no logs added, CL = cultivated with logs added, CN = cultivated with no logs added). Lower case letters indicate significantly different mean health among treatments within each individual site.

6.4 Discussion

6.4.1 Site

The strongest influence on the survival of planted seedlings over a year was the site at which it was planted. The condition of a research site prior to planting, as measured by structural complexity, was strongly associated with the amount of planted eucalypt seedlings that were alive a year after planting. This supports the proposal from state and transition models of degrading processes in remnant vegetation that as degradation increases, reversal to a less degraded state becomes progressively more difficult (Yates and Hobbs 1997b; McDonald 2000; Suding *et al.* 2004; Spooner and Allcock 2006).

Suding *et al.* (2004) also proposed that the dynamics of degraded sites may differ from those of less modified sites. In the two locations that had both healthy and degraded sites (Epping Forest and Fingal) planted seedlings in the healthy sites had significantly greater chance of surviving than in the degraded sites. Species and treatment effects on seedling survival, relative growth and health also differed between healthy and degraded sites and thus support this proposal. These results also imply that the restoration treatments did not completely relieve the existing constraints to regeneration at the degraded sites and implicates localised biotic and abiotic conditions rather than broader climatic ones alone.

Particularly high survival of seedlings occurred at the Evercreech coupe site in all treatments. This shows that the soil amelioration techniques used were capable of promoting successful seedling establishment. The coupe site was a particularly resilient one however, and would not normally need intervention to maintain healthy plant and microbial populations as conditions were relatively benign and natural

regeneration was present across the site. This is in stark contrast to the adjacent remnant enclosed by plantation which had suffered high levels of modification due to previous agricultural practices (Close *et al.* 2008). Here the establishment treatments were less successful. The seedlings had 40% greater mortality than the coupe site with a pattern of substantial mortality soon after planting and relatively few deaths over summer. One remarkable difference between the two sites was that of soil water repellency with the soils in the coupe site generally having very low levels of background water repellency and the plantation site having soils with that were intrinsically severely to extremely hydrophobic (see Table 5-8).

In contrast, the two sites at Epping Forest both had extreme levels of water repellency and they experienced similar trends in seedling mortality over the year. Mortality was still amplified at the degraded site Valleyfield, however. For a degraded site, Valleyfield had relatively few early deaths post planting but went through an extreme morality event between mid December 2008 and mid January 2009. The Tom Gibson Reserve site also had a drop in survival over the same period and both sites only had 5-6% mortality for the rest of the year.

At Oatlands, where the two sites were of similar structural complexity, the site that had the most eucalypt recruitment from seed at the end of two years (Bald Hill, see Chapter 5) had the highest levels of water repellency and lowest overall survival rate of planted seedlings. These were both significantly different to Liliesleaf. The seedlings at Bald Hill suffered the most severe case of transplant shock of all the sites as almost half died within a month of planting while Liliesleaf lost around a quarter of the seedlings over this time. The extent of these early deaths may have been heightened by the later planting times at these sites relative to others as the soils may have been considerably drier.

The Oatlands sites were unavoidably planted in mid October (2-3 weeks later than other sites) and conditions were drier and warmer but frosts were still occurring, with minus 3.6°C experienced a week after planting. However, the seedlings were well hardened in the nursery and would have experienced numerous frosts at the nursery prior to planting (pers. comm. Tony Waits Woodlea Nursery manager) suggesting that additional factors to climatic ones may have contributed to mortality.

Soil water repellency is implicated in all the mortality patterns described above and will be discussed further in section 6.4.4. A factor that was not measured directly in this study that may have implications for differential seedling survival among sites is soil nutrients (Close *et al.* 2008). Nutrient enrichment in degraded sites may disadvantage planted eucalypt seedlings and possibly cause early death. Other eucalypt restoration studies found that fertilising newly planted seedlings in pasture sites decreased the survival and growth of eucalypt seedlings (Li *et al.* 2003; Semple and Koen 2003) and fertilising seed beds decreased the germination and survival of eucalypts (Pinkard 1992; Skinner *et al.* 2010). Competition from vigorously growing grasses and exotic herbs is often implicated.

Differences in nutrients at different sites, their interaction with fire and with understorey species may also have consequences for mycorrhizal fungal communities (Ellis and Pennington 1992; Launonen *et al.* 1999; Tommerup and Bougher 2000). Consequently this may effect the growth and survival of planted seedlings in different ways at different sites (Teste *et al.* 2009b). Further studies including nutrient analyses would help understand these effects.

6.4.2 Species

Different species and sub-populations within species have different capacities to acclimatise to the changes in environmental parameters such as irradiance, temperature, frost and drought (Battaglia *et al.* 1996; Davidson *et al.* 2004; Close *et al.* 2009b) that occur when seedlings are transferred from a nursery to a field environment (Close *et al.* 2005a). In addition different strategies for coping with low soil water and high atmospheric evaporative demand during summer may contribute to differences between species in competitive ability and survival (Close *et al.* 2009b). This may help explain the much stronger effect of species than treatment on the likelihood of planted seedlings dying in this study. Survival of *E. viminalis* over a year was significantly greater than *E. amygdalina* at all sites, except the healthiest site Evercreech Coupe (where survival was similarly high), and was largely unaffected by establishment treatment, except at Liliesleaf at Oatlands where *E. viminalis* survived better in cultivated plots than burnt plots. Relative growth of surviving seedlings at the end of the year since planting was not significantly different between species however.

The two species used in this study naturally co-occur in the surrounding forest at Epping Forest and Fingal Valley with *E. amygdalina* usually being the dominant species. *E. amygdalina* is present but not common at Oatlands where *E. pauciflora* is the dominant and *E. viminalis* subdominant (Kirkpatrick and Gilfedder 2000; Harris and Kitchener 2005). However, while overall survival of *E. amygdalina* was low in the first year at the two Oatlands sites (18.2 and 34.7 %) it was still higher than at the degraded Epping Forest site Valleyfield (11.9%). This continued into the second year as there were relatively few seedling deaths at these three sites during that time.

Close *et al.* (2010a) concluded from their study on planting a range of local and non local eucalypt species on ex pasture sites in the Midlands, that seedling ecophysiological traits are key to success in highly modified agricultural landscapes. While no specific physiological measurements were made in the current study, reasons why these two species differed so dramatically may be speculated upon. The two species are in different subgenera with *Eucalyptus viminalis* being in the subgenus *Symphyomyrtus* and *E. amygdalina* in *Eucalyptus* (previously *Monocalyptus),* and differing traits between subgenera may suggest reasons for the differential in survival.

Davidson and Reid (1980) showed in glasshouse studies that members of the subgenus *Monocalyptus* established more slowly from seed and had a lower root to shoot ratio than those of *Symphyomyrtus.* In a paper comparing the ecological traits of the two subgenera Noble (1989) suggested that *Monocalyptus* species: tend to occur on more mesic sites and at sites where they do co-occur suffer greater damage during droughts; are less resistant to frost; tend to be found on soils of lower nutrient availability; appear to be more dependent on mycorrhizae for vigorous growth; show slower germination, resprouting and early growth than *Symphyomyrtus* species but once established have relatively rapid growth rates.

After analysis of a database of nutrient concentrations in components of 110 eucalypt species, Judd et al.(1996) found that concentrations of calcium, manganese and potassium are lower and magnesium greater in *Monocalyptus* leaves, bark and wood than in *Symphyomyrtus*. Concentrations of nitrogen and phosphorus were also higher, but not significantly, in *Symphyomyrtus*. This may lead to *Monocalyptus* species being more prone to toxicity reactions in elevated nutrient conditions (Chambers and Attiwill 1994), such as those in previously intensely grazed sites (Close et al. 2008),

and particularly in soils of elevated nutrition that also get intensely burnt to an ashbed. As more *E. amygdalina* died within the first weeks of planting in burnt plots than in cultivated plots and *E. viminalis* was not so severely affected this is a possible reason for early differences in survival between the species. As this was not directly tested in this study, it would be an interesting line of inquiry for further studies.

While *Eucalyptus viminalis* seedlings survived better than *E. amygdalina*, it was more susceptible to herbivory by insects and this was reflected in generally lower health scores at most sites. This study has shown therefore, that choice of species is a critical decision that needs to be considered when undertaking restoration plantings. Very different outcomes can occur solely due to the species planted.

6.4.3 Treatments

With data from all sites and species combined, the treatment that had the highest probability of planted eucalypt seedlings surviving over the year was in cultivated without log plots. This indicates that planted seedlings, most particularly in degraded sites, tended to survive better in conditions least similar to the natural recruitment niche described in Chapter 3.

This result is in agreement with Li *et al.* (2003) who found, in a fully factorial experiment looking at the effects of grazing exclusion, shallow and deep cultivation, burning and fertiliser addition on the survival and growth of *Eucalyptus blakelyi* seedlings in the NSW tablelands, that the best results were achieved when grazing was excluded and plots were deep (20cm) cultivated. It also aligns well with the techniques suggested as current best practice for establishing seedlings in ex pasture sites by Close and Davidson (2003) and confirmed by Close *et al.* (2010b). In a six year study on the effects of a range of additional establishment methods on the

growth and survival of the local species *Eucalyptus pauciflora* and non local *Eucalyptus nitens*, none of the additional techniques (including spot burns) significantly increased survival and growth at all experimental sites above the control of current best practice techniques on their own (Close *et al.* 2010b).

The hazard of seedling mortality changed over time in different treatments. Spot burns as an establishment treatment resulted in significant death of seedlings very soon after planting in degraded remnants, particularly so at Oatlands and most severely in *E. amygdalina*. However, seedlings in cultivated plots, particularly those with logs, underwent a marked decrease in survival over summer and ended up having lower survival than burnt plots at some sites (including Tom Gibson *E. amygdalina*). This may reflect changes in water availability due to increasing competitive effects as weeds and grasses grew and/or re-establishment of inherent water repellency as soils dried out.

As discussed in the previous chapter, cultivation as an establishment treatment reduces soil strength which decreases the amount of energy a seedling has to expend to extend its roots into the soil profile (Close *et al.* 2005a), enabling greater penetration of the soil for water in cultivated soil than in uncultivated and/or burnt soil (Passioura 1991; Skinner *et al.* 2009). Cultivation can also, however, increase the incidence of weeds which, if not managed effectively, causes competition for water and nutrients (Close and Davidson 2003). This is one plausible explanation of why there was a crash in survival over summer in cultivated plots. Why plots with logs were more severely affected at that time is unclear but may be due to greater lodgement of weed seeds next to logs and therefore greater establishment of competitors and/or dry logs may have absorbed the light rainfall and prevented water from reaching the soil.

At the end of two years, as found with seedlings recruited from seed, the addition of logs to plots did not necessarily benefit or inhibit the survival of planted seedlings of both species. The benefits that coarse woody debris provides recruited seedlings in a more natural situation may have been negated by the restoration methods used, as described in Chapter 5. The provision of a moisture store for newly planted seedlings under logs in particular may have been more effectively provided if the logs had been placed on the plots for a longer time before planting or if plots had been established around existing coarse woody debris. Further research into the use of logs as a restoration method is warranted.

Growth of seedlings was generally poorer in cultivated than burnt plots and this too may be due to increased competition (Skinner *et al.* 2010). Burnt plots generally maintained lower levels of weeds (as shown in the previous chapter) and this, plus a likely post fire pulse of nutrients (Chambers and Attiwill 1994) would have had beneficial effects on seedling growth for surviving seedlings. The most marked differences in seedling growth in burnt and cultivated plots occurred in the healthy sites Tom Gibson and Evercreech Coupe. This suggests that fire effects are influenced by the initial soil condition at a site and implicates nutrition (Certini 2005). The healthier sites would have had lower levels of nitrogen and phosphorous than the degraded remnants that had histories of grazing and were adjacent to fertilised paddocks (Close *et al.* 2008; Duncan *et al.* 2008) and thus showed a more positive response to the increased availability of nutrients in ashbeds. Additionally, herbicide treatments were not undertaken in cultivated plots at the healthy sites as there were few exotic species in the reserves. At Tom Gibson Reserve in particular, native grasses, small shrubs and bulbous plants regrew well in the cultivated plots

and may have competed strongly with the eucalypt seedlings for limited nutrients and water, especially over summer.

The majority of degraded remnants (except Liliesleaf) also showed some evidence of more rapid growth in ashbeds compared to cultivated plots. This shows that for seedlings that survived the amplified transplant shock experienced in burnt plots, the growth benefits were similar to those of the ashbed effect reported in logged forests (Pryor 1963; Renbuss *et al.* 1973) and may give a long term growth advantage (Forestry Commission 1993). Indeed the exact phenomenon of increased early mortality but much improved growth in survivors in intensely burned ashbeds relative to unburnt soils has been sporadically reported in forestry literature since the 1950's (cited in the Forestry Commission (1993) booklet). One cited source (De Bano and Rice 1973) attributes the early mortality of seedlings established from seed in ashbeds to drought resulting from a water repellent soil layer. This also has implications for seedlings planted into ashbeds.

6.4.4 Soil water repellency

Soil water content and hydraulic properties of the soil system can affect whether stress occurs in planted seedlings (Grossnickle 2005). Both these factors are influenced by the spatial and temporal heterogeneity of soil water repellency (Dekker and Ritsema 1994; Doerr *et al.* 2000; Leighton-Boyce *et al.* 2005). In this study the survival of planted seedlings over time was significantly related to the inherent soil water repellency of the plots in which they were growing. With every increase in soil water repellency class there was a 20% increase in the hazard of a seedling dying during the year after planting. This link between soil water repellency and planted seedling survival has not been widely reported since De Bano and Rice (1973) but

has important implications for the success of restoration of eucalypt woodlands and forests with naturally water repellent soils.

According to De Bano and Rice (1973), when seedlings are planted by hand into soil that has a fire induced water repellent layer "the water-repellent layer is penetrated by the planting hole, but water repellency in the surrounding area prevents or greatly reduces infiltration". As a result the moisture available to the seedling may be quite small. This is most likely to have occurred in this study, particularly at the site with the most hydrophobic soils, Bald Hill, which experienced the most severe case of seedling transplant shock, especially in the burnt plots.

When seedlings were watered at planting, the water was observed to infiltrate rapidly in burnt plots. This was most likely due to the water moving horizontally across the fire- induced water repellent layer as distribution flow (DeBano 2000; Doerr *et al.* 2000) with some limited vertical movement into the seedling root ball. The majority of the soil surrounding the seedling root ball would have therefore remained substantially dry and water repellent. In addition, coarse-textured potting media can lose moisture to finer-textured or drier soils (Heiskanen and Rikala 2000; Close *et al.* 2005a) and can also become hydrophobic when dry (Handreck and Black 2002). This process would have exacerbated the drought stress to seedlings and could have been instrumental in early mortality.

Results reported in Chapter 5 showed that cultivated plots could also be inherently water repellent at both degraded and healthy sites and that they were generally no less hydrophobic than burnt plots. Once the moisture from initial watering had been utilised by the seedlings, dry weather over summer would have caused the soil surrounding the root ball and the potting mix around the roots to dry out. Their

hydrophobic nature would resist rewetting and could cause death of seedlings for some time after planting. This may also influence the growth of seedlings over time as suggested by a significant correlation between relative growth of surviving seedlings and plot level soil water repellency (WDPT).

Soil water repellency in eucalypt woodlands and dry forests is a phenomenon with important implications for the success of restoration efforts that has not previously been investigated in this context. Planted seedlings with limited root ball volume would be expected to be more affected by water repellency in adjacent soil than naturally recruited lignotuberous sprouts that have ten times the root volume (Burdett 1990) and well developed storage organs (Bell and Williams 1997). Future work could include the trialling of wetting agents and gels with the aim of improving seedling establishment and growth as has been undertaken in other environments (Osborn *et al.* 1967; Dekker *et al.* 2005a; Madsen 2010).

6.4.5 Remnant structural attributes

In addition to the overall structural complexity of a site, seedling survival at the end of a year after planting was significantly correlated with two individual structural attributes: stand basal area and the number of large trees (dbh<40cm). This is in contrast to most forestry literature which suggests that seedling survival decreases with increasing retention of "overwood", particularly in dry forests where moisture is limiting (Bowman and Kirkpatrick 1986; Battaglia and Wilson 1990; McCormick 1991). Competition for moisture, nutrients and possible allelopathic affects have been given as possible causes of the suppressive effects of overwood on regeneration in dry forests (McCormick 1991) and in Western Australian woodlands (Yates *et al.* 1996; Yates *et al.* 2000a).

However, in degraded woodland situations an increased number of large trees may have beneficial microclimatic effects by modifying extremes of wind, temperature and frost. In more open degraded sites exposure to wind is increased. This dries the soil surface and may cause increased plant moisture loss through increased evapotranspiration (Close and Davidson 2003) and thus induce planted seedling mortality. Seedlings in more open areas are also more subject to damage from frost (Bowman and Kirkpatrick 1986). The time taken to establish mycorrhizal associations and thus the proximity to a source of fungal infection may also be important in the survival of seedlings. With a larger number of large trees seedlings may be able to link into mycorrhizal networks more easily (Teste *et al.* 2009a).

On a smaller scale, however, this study showed a significant relationship between seedling survival and distance from closest tree at all but one site measured. The further away a seedling was from a tree the more likely it was to survive over the year. This effect is reported to also occur in healthy dry forests with regeneration been shown to be suppressed within an area of up to two crown radii around the trunk of large trees (McCormick 1991). Yates *et al.* (2000a) also reported that the presence of adult *Eucalyptus salmonophloia* had a clear negative effect on the establishment of seedlings of the same species in degraded woodlands. This was attributed to the adults being efficient harvesters of water.

These results show that the placement of restoration treatment patches within remnant stands needs careful consideration. A balance needs to be struck so that planted seedlings benefit from the greater protection from environmental extremes afforded by being surrounded by large adult trees but avoid restrictive competition for resources, particularly water. Experiments more thoroughly trialling spatial aspects of eucalypt regeneration may elucidate these issues further.

6.5 Conclusions

This study has shown that the early establishment of planted eucalypt seedlings in woodland remnants is significantly affected by the condition of the remnant; the species used (and possibly whether the species is from *Eucalyptus* subgenus *Eucalyptus* or *Symphyomyrtus*); the underlying soil water repellency of the planting plot; the proximity to an adult tree and the type of soil amelioration used in the restoration treatment but not by the presence or absence of logs. While there were a number of complex interactions between site, species and establishment treatments, overall:

- healthy sites had significantly better establishment of seedlings than nearby degraded sites;
- degraded sites had more severe incidence of transplant shock and early death of seedlings and/or amplified death during summer compared to healthy sites;
- *Eucalyptus viminalis* (from subspecies *Symphyomyrtus*) performed significantly better at most sites and was less affected by establishment treatments than *Eucalyptus amygdalina* (subspecies *Eucalyptus*) possibly due to differences in ecophysiological traits such as more rapid root growth and avoidance of drought, drought tolerance and susceptibility to toxicity in nutrient enriched sites (Noble 1989; Close et al. 2005a);
- relative growth of surviving seedlings after a year were similar between species and was generally higher in burnt plots;
- surviving *E. amygdalina* were generally healthier than *E. viminalis*;
- seedlings planted in cultivated plots without logs (persistence niche analogue) had a greater chance of surviving over a year than seedlings in burnt with log plots (recruitment niche analogue);

- the hazard of a seedling dying changed over time in different treatments with seedlings in burnt plots more likely to die soon after planting and seedlings in cultivated with log plots more likely to die over summer after which survival was similar in all treatments up to two years;
- seedling survival over a year was significantly affected by structural and spatial attributes such as the amount and size of trees in a remnant and the distance a seedling was from them and;
- soil water repellency had a significant influence on the survival of planted seedlings over a year.

The complexity of these results reflects the heterogeneity of the woodland systems overlaid with the further unpredictability of the effects of weather and of degrading influences such as grazing and nutrient enrichment in these dry agricultural areas (Yates *et al.* 2000a). There appears to be tradeoffs between survival and growth/ health in seedlings of different species and in different treatments. It is thus difficult to make broad management prescriptions aimed at optimising eucalypt regeneration and recreating the persistence niche across a range of sites.

However, this study has provided focus for further experimental work that may help further refine these restoration techniques and elucidate relationships between ease of regeneration and soil, competition and spatial characteristics. Issues that need further clarification include: the timing and intensity of weed control in cultivated treatments; the fuel load, spatial arrangement and intensity of fire in burn treatments; the benefits of a combined cultivation/burn treatment; the possible toxicity effects of fire in nutrient enriched soils; the effects of soil water repellency amelioration through the use of wetting agents; and spatial aspects of treatment patch placement.

A unique outcome of this study has been the highlighting of the effects of soil water repellency on planted seedlings survival and its implications for the success of restoration treatments in dry eucalypt woodlands.

Chapter 7 General discussion

Lack of eucalypt regeneration is a key factor in the decline of forest and woodland remnants in low rainfall agricultural regions in Australia (Yates and Hobbs 1997a; Gibbons *et al.* 2008b; Fischer *et al.* 2009). Spooner and Allcock (2006) proposed that development of appropriate management and restoration measures in these systems requires a better understanding of the ecology of regeneration and the development of appropriate techniques for restoration.

This thesis has significantly advanced the ecological understanding of the eucalypt regeneration process in Tasmanian dry forests and woodlands by quantifying eucalypt regeneration in stands of different condition and disturbance history; identifying which structural elements of the ecosystem the quantity of regeneration is correlated with; describing the elements of the natural seedling recruitment niche; documenting ontogenetic niche shifts; and highlighting the importance of soil water repellency heterogeneity. Improved insight has also been gained of the early dynamics of eucalypt regeneration in woodland remnants in different disturbance conditions following restoration treatments that attempted to mimic the eucalypt recruitment and persistence niches.

This final chapter summarises the principal results of previous chapters in this thesis. These results are then synthesised in a conceptual model of woodland/dry forest regeneration states, in an appraisal of restoration techniques and in a discussion of the implications of soil water repellency. Research gaps are identified and suggestions made for the management of woodland and dry forest remnants in order to improve the quantity and quality of eucalypt regeneration in the Midlands of Tasmania.

7.1 Principal research results of the thesis

Stand scale characteristics associated with the quantity of eucalypt regeneration

The study described in chapter 2 investigated eucalypt regeneration at a stand scale through the quantification of structural complexity in thirty woodland and dry forest remnants in a range of conditions. No other study of this type has been undertaken in the Tasmanian Midlands.

The quantity of eucalypt regeneration was positively correlated with perennial species and life form richness, cover of vegetation 0.5-6m high, dry litter weight, the numbers of dead trees, the number of large trees, the total length of fallen logs >10cm diameter and negatively correlated with quadratic mean diameter of trees on the sites. The quantity of eucalypt regenerating stems was also shown to be degraded in parallel with the overall condition of the woodland remnants with four times the amount of eucalypt regeneration found in healthy sites than in degraded sites and most often none in paddock tree stands. In agreement with studies elsewhere, grazing history was shown to influence the quantity of eucalypt regeneration within remnants. Sites with a history of light to moderate livestock grazing had similar quantities of regeneration to each other but these sites had significantly more regenerating stems than sites with histories of heavy and extreme grazing.

The importance of fire in the eucalypt regeneration process was highlighted in this study in that woodland remnants that had been burnt within the past ten years had greater quantities of eucalypt regeneration than unburnt sites. With the exception of one unburnt site, only burnt sites contained regeneration in the form of seedlings.

An important outcome of this study was the revelation that three quarters of remnants studied had long term regeneration deficits and did not have sustainable, population-replacing quantities of eucalypt regeneration.

The eucalypt recruitment and persistence niches in Tasmanian dry forest and woodlands

The study described in Chapter 3 investigated eucalypt regeneration at the microsite scale through a survey conducted in four healthy naturally regenerating woodland/open forest remnants. This study extended the knowledge of eucalypt regeneration in Tasmanian dry forests and woodlands by identifying the important attributes associated with natural eucalypt recruitment and juvenile persistence niches. An ontogenetic niche shift was shown to occur as the attributes of the recruitment and persistence niches were shown, on the whole, to be significantly different.

The recruitment niche was shown to be tightly circumscribed. Eucalypt seedlings were predominantly found in microsites that were in canopy gaps, in ashbeds, sheltered to a large extent by coarse woody debris and in soil that was softer and less water repellent than the surrounding forest floor. This was in contrast to the juvenile persistence niche. The microsites that contained lignotuberous sprouts had shelter, vegetation cover, ground cover and soil attributes similar to those of the surrounding forest floor with the exception of softer soil and being predominantly located in canopy gaps.

Soil water repellency heterogeneity and implications for eucalypt recruitment

The study described in Chapter 4 was located in two recently burnt woodland remnants and further explored the soil moisture and water repellency characteristics of typical eucalypt seedling microsites (in ashbeds near coarse woody debris) and of the surrounding forest floor. This is the first study to describe the spatial heterogeneity of soil water repellency in this context and to relate it to the eucalypt recruitment process.

The soil water repellency characteristics of typical eucalypt seedling microsites showed the classic pattern of soil heated under high intensity fire. In ashbeds adjacent to partially burnt logs, soil was wettable for 1-2 centimetres below the surface and this wettable layer was underlain by a severely water repellent layer of at least one centimetre thickness with less repellent soil below that. This layered pattern generally occurred horizontally across ashbeds up to a distance of a meter from the central log, with some patchiness. In areas of less intense burn outside ashbeds and in unburnt soil, the surface layers were moderately to extremely water repellent.

Soil under burnt and unburnt logs contained three times more moisture in the top 10cm than the soil in areas five metres away from logs. This represents a significant accessible soil moisture store for seedlings adjacent to logs.

Surface soils in ashbeds sampled after substantial rainfall were shown to have higher moisture contents than the repellent layer below, with even less moisture in the layer below that. High moisture contents in potentially severely water repellent surface soils outside of ashbeds revealed the potential of inherent soil water repellency to act as a barrier to the transition between seed germination and seedling establishment.

Seeds that germinate in soils that are moist during winter-spring wet seasons but dry to become severely water repellent could become cut off from soil moisture during dry summer periods.

The patterns of soil water repellency within ashbeds and the differences between burnt and unburnt patches revealed in this study all have implications for seedling establishment. This study thereby highlights the presence of soil water repellency as a potential constraint on eucalypt recruitment.

Patch scale restoration of the recruitment niche

The study described in Chapters 5 identified potential barriers to eucalypt recruitment and analysed the effects of restoration treatments on germination and survival of seedlings from manually and self-sown seed in six remnant woodland stands of varying condition. The treatments aimed at mimicking elements of the natural recruitment and persistence niches as described in Chapter 3 through the use of intense spot burns or cultivation and the addition of logs.

Cultivation and burning treatments each provided a seedbed for hand sown seed and natural seed rain that was superior to untreated ground, implying that some of the barriers to eucalypt recruitment had been removed, at least to some extent. However the designated analogue to the natural recruitment niche (burn with log treatment) did not significantly improve recruitment compared to other treatments. There was little consistency in the treatment effects on survival of seedlings from manually sown and self-sown seed.

There were significantly more germinants of hand sown seed in healthy sites than in degraded sites and the amount of observed germination was significantly correlated with the structural complexity of a site and with the total length of logs at a site.

However the number of surviving seedlings a year after sowing did not correlate with site condition and there were no significant differences among sites. At all sites, less than 1% of estimated viable seed sown resulted in established seedlings one year after sowing.

The need for good weed control was confirmed by a significant positive association between the amount of established seedlings and the extent of bare earth and a negative association with the cover of herbaceous weeds.

The numbers of seedlings established at the end of two years at three degraded sites were limited. Results from this study confirmed that establishing eucalypt regeneration from sown seed is influenced by a multitude of factors including a strong stochastic element and therefore artificially recreating the recruitment niche is not an easy task.

Patch scale restoration of the persistence niche

The study described in Chapter 6 was undertaken at the same sites and during the same period as those described in Chapter 5. Seedlings of two local species of eucalypts were planted at the six research sites in four establishment treatments to assess whether planted seedlings established and performed best in conditions aiming to mimic those where eucalypts naturally recruit or where they persist.

Seedling survival over a year was significantly correlated with the condition of a site as measured by structural complexity. Seedling survival was significantly higher in healthy sites than nearby degraded sites. This indicates that the persistence niche is more readily sustained in less disturbed sites. More severe incidence of transplant shock, early mortality and/or amplified mortality of seedlings during summer was reported at degraded compared to healthy sites.

The species of a seedling had a major effect on its probability of surviving over a year. Significantly more *Eucalyptus viminalis* (subgenus *Symphyomyrtus*) seedlings survived at most sites and was less affected by establishment treatments than *Eucalyptus amygdalina* (subgenus *Eucalyptus*), most likely due to differences in ecophysiological traits such as drought tolerance (Noble 1989; Close *et al.* 2005a). Surviving *E. amygdalina* seedlings were generally healthier than *E. viminalis* seedlings, however. Relative growth of seedlings surviving after a year was similar for both species and was generally higher in burnt than cultivated plots.

Seedlings planted in the designated persistence niche analogue treatment i.e. cultivated without log additions, had a greater chance of surviving over a year than seedlings in the designated recruitment niche analogue, burnt with added logs. The hazard of a seedling dying changed over time in different treatments. Seedlings in burnt plots were more likely to die soon after planting and seedlings in cultivated plots with added logs were more likely to die during summer. Survival was similar in all treatments during the second year with few seedlings dying at the three degraded sites assessed.

Soil water repellency had a significant influence on the survival of planted seedlings during the first year as did the amount and size of trees in a remnant and their distance from a seedling.

7.2 Conceptual model

State and transition models are used here (see Fig 7.1) to provide a conceptual framework in which to place the results of the above research results, to identify research gaps and to provide guidance for management of woodland and dry forest remnants in the Midlands of Tasmania. The following discussion incorporates factors that were not the subject of experiments conducted in this thesis but have been shown in other studies to play a vital role in restoration success. This provides a more holistic picture of the complex interactions involved in restoration of degraded dry forests in Tasmania and offers a suitable context for the management application of the main findings outlined in this thesis.

In Chapter 2, the study of structural complexity at a stand scale showed that regeneration pools in Tasmanian woodland and open forest remnants tended to diminish in parallel with the decline in overall structural complexity and health condition of remnant vegetation. Only a limited number of study sites contained regeneration at population-replacing levels (i.e. regenerating stem to adult ratio >1:1). In the state and transition model below, a matrix of forest/woodland condition by amount of regeneration has been used to divide remnant woodlands into eight states (S1- S8 in Figure 7-1) ranging from healthy with plentiful regeneration (S1, regeneration to adult ratio >2:1), through intermediate with limited regeneration (S5, regeneration to adult ratio < 1:1) to poor paddock tree sites with no regeneration (S8).

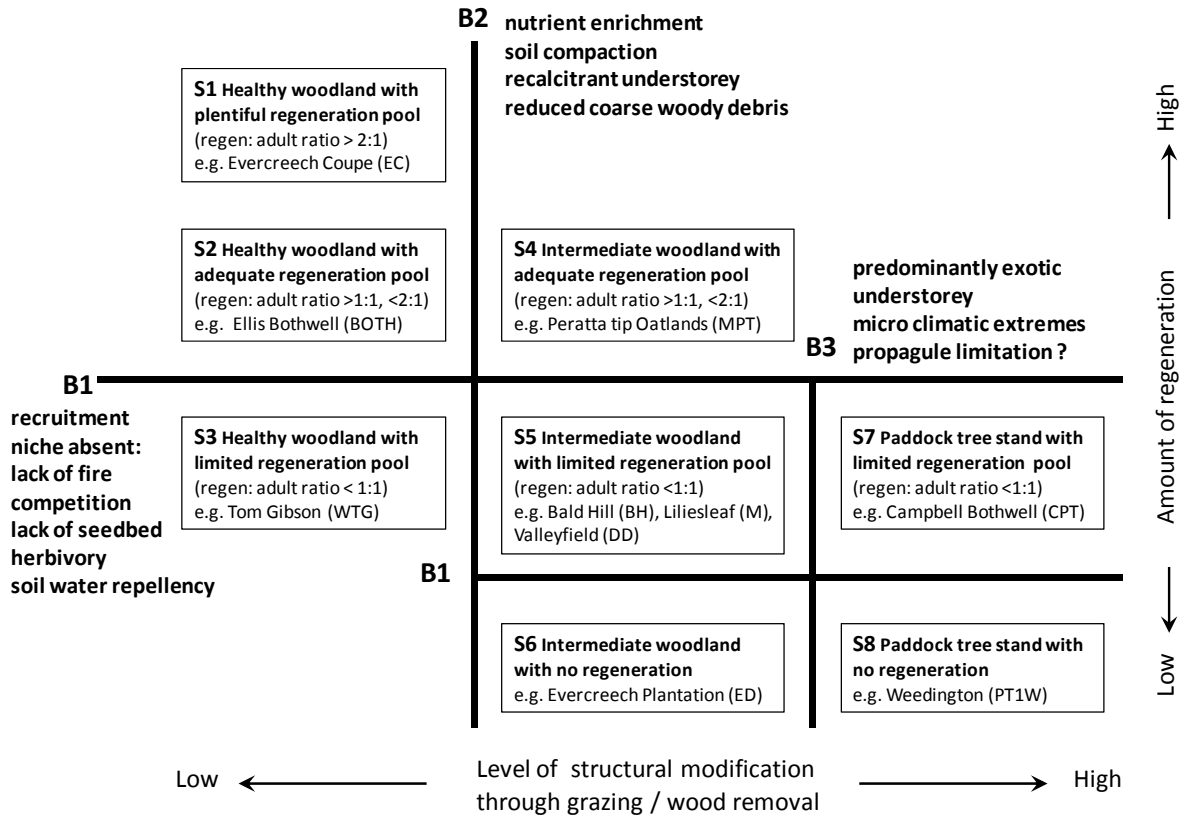

Figure 7-1 A conceptual model of remnant woodland (and dry open forest) states in the Midlands of Tasmania with respect to eucalypt regeneration and structural modification and showing barriers to transitions between states. Boxes represent states (S1-8) each of which is a unique combination of remnant condition (healthy, intermediate, paddock tree) and level of regeneration (prolific, adequate, limited or none). Regeneration to adult ratios and example of sites assessed in Chapter 2 (abbreviations in brackets) are shown. Heavy lines (labelled B1-3) represent barriers to transition between states. Biotic and abiotic constraints that may contribute to these barriers are listed in bold type.

Reduction of structural complexity of remnant vegetation through management practices such as grazing, clearing, wood collection and either over use or recurrent suppression of fire leads to degradation of regeneration pools (Chapter 2; Fischer *et al.* 2009). In addition, long term fire suppression alone can result in limited regeneration in healthy remnant stands (Yates *et al.* 1994b; Close *et al.* 2009a). This was observed to be the case in the reserves assessed in Chapter 3. These factors in concert with natural phenomena identified in Chapter 5 such as limited and variable rainfall, frost, inherent soil water repellency and competition from adult trees and

understorey species, preclude the eucalypt recruitment niche identified in Chapter 3 and constitute a barrier (B1 in Figure -1) to transition from limited to adequate regeneration. Lack of recruitment niche attributes and the resultant absence of eucalypt seedling recruitment may therefore occur in stands of good (S3), intermediate (S5-6) and poor (S7-8) condition.

The aim of the restoration trials reported in Chapters 5 and 6 was to assess whether the techniques based on the recruitment and persistence niches described in Chapter 3 reduced the barriers to regeneration (B1 in Figure 7-1) and assisted sites to a state of improved recruitment. The results suggested that the regeneration dynamics of sites in different states do indeed differ after restoration. On the whole, the effectiveness of restoration methods differed significantly between sites which were in intermediate versus healthy states, particularly for planted seedlings. This was most likely due to the identified barriers between healthy and intermediate sites (B2 in Figure 7-1) interacting with the barriers to regeneration (B1). The effects of water repellency and importance of species choice on the success of seedlings planted at sites in different states were highlighted. The effects of other elements described in the B2 barrier (e.g. nutrient enrichment, soil compaction, recalcitrant understorey), while studied in woodlands elsewhere (Yates *et al.* 2000b; Prober *et al.* 2002; Spooner and Allcock 2006; Fischer *et al.* 2009), need further elucidation in the Tasmanian context.

The exact nature and effects of the barriers preventing transition of paddock tree sites back to intermediate condition (B3 in Figure 7-1) were also not directly studied in this thesis. However, work elsewhere shows that these barriers do exist as the removal of the degrading processes (i.e. heavy grazing and /or cultivation and

fertilisation) has been shown not to be sufficient to encourage spontaneous recovery or regeneration (Prober *et al.* 2002; Spooner *et al.* 2002; Li *et al.* 2003; Suding *et al.* 2004; Cramer *et al.* 2006; Standish *et al.* 2007; Spooner and Briggs 2008). Techniques to restore trees back into ex pasture sites in the Tasmanian Midlands have also been well studied (Pinkard 1992; Close and Davidson 2003; Churchill 2004; Close *et al.* 2005b; Nardon *et al.* 2005; Close *et al.* 2010a; Close *et al.* 2010b). It was also of interest, therefore to test how well these techniques could be adapted to restoration within remnant woodland patches.

7.3 Appraisal of restoration techniques and suggestions for further research

Other studies have shown that establishing eucalypt regeneration in remnants with a grassy understorey and in grazed land is extremely difficult (Ellis and Pennington 1992; Fensham and Kirkpatrick 1992; Pinkard 1992; Clarke and Davidson 2001; Clarke 2002; Li *et al.* 2003; Semple and Koen 2003; Skinner *et al.* 2009; Orscheg *et al.* 2011). The results from the restoration trial reported in Chapters 5 and 6 confirmed this with low numbers of new recruits at some sites and less than 50% survival of planted seedlings at three degraded sites two years after planting. However the recruitment niche was enhanced at all sites and indeed the niche space was broadened (as described by Young *et al.* 2005). This was evidenced by similar levels of recruitment being achieved in cultivated plots as in burnt plots with logs added (recruitment niche analogue).

High seedling survival rates were achieved after one year for each of the two eucalypt species when planted in the healthiest forest remnant, showing that the techniques used could produce good results. Regeneration following seeding was

generally less successful than via planted seedlings. Direct seeding of experimental plots placed in thick ash beds following burning resulted in no germination of sown seed. Self-sown seeds did germinate and natural regeneration eventually established on the edge of the ashbed. Therefore it is suggested that seed be placed on the edge of the ashbed in sites that have undergone restoration using a hot burn with large piles of fuel (e.g. where woody weeds are killed, heaped and burned).

Both cultivation and intense spot burns had drawbacks and benefits as establishment treatments in sites of intermediate health and this may have been in part due to the techniques used. The spot burn technique attempted to mimic the large scale disturbance of fire on a small scale so as not to endanger whole remnant populations of trees (as suggested by Yates *et al.* 1994). This technique also mimics the prescribed burning of 'turkey heaps' used by Forestry Tasmania to produce ash beds to promote regeneration in selectively harvested native dry forest coupes (Forestry Commission 1993). However the limited germination and poor survival of planted seedlings in burnt plots relative to cultivated plots in degraded sites suggested that this technique did not successfully recreate the recruitment niche found in naturally regenerating remnants.

Lack of heterogeneity of soil water repellency due to a relatively uniform intensity of fire and lack of heterogeneity of microtopography in the seed bed due to soil compaction may have affected microsite moisture availability for seedlings in burn treatments (Chapter 4). This in turn explains the limited germination and severe transplant shock for planted seedlings at some degraded sites. In order to refine the spot burn technique, further research is needed on the desired fuel load, spatial arrangement and intensity of fire in burn treatments and on the potential benefits of a combined cultivation/burn treatment.

However, the burn technique did provide longer term and more effective weed control than the cultivated treatment and surviving seedlings benefited through improved growth and health from the ashbed effect at most sites, particularly sites within healthy forest remnants. As weed control is essential for restoration in degraded sites (Pinkard 1992; Yates *et al.* 2000a; Close and Davidson 2003) and can be a costly process that requires regular follow up management (Venning 1988; Close and Davidson 2003), the demonstrated weed control benefits of fire are considerable. Similarly, good early growth of seedlings helps them to occupy a site quickly and to dominate the competition for resources, therefore decreasing the need for such intensive weed control (Forestry Commission 1993; Adams et al 2003). These considerable benefits need to be weighed up against possible increased early mortality when contemplating using spot burns as a restoration technique. Again, further research may elucidate ideal fuel loads and intensities that provide these benefits while also improving germination and survival rates of seedlings.

Cultivation provided the best early seed bed and early survival of planted eucalypt seedlings. However the combination of increased competition from exotic species and effects of inherent soil water repellency on water availability resulted in increased mortality and reduced growth of seedlings through the first summer. The benefits of using this herbicide treatment did not appear to outweigh the negatives as the weed control was not complete. More effective weed control may have been achieved in cultivated plots with a pre-sowing application of herbicide (in addition to initial glyphosate application prior to cultivation) rather than, or as well as, post planting (Florence 1996; Close and Davidson 2003; Nardon *et al.* 2005). This would be a strong recommendation in future restoration efforts if and where the use of fire

as an establishment treatment is not possible. Further research into the timing, type and intensity of weed control is warranted.

Castro et al. (2011) suggest that coarse woody debris could be used to protect regenerating seedlings, with potential for use in the restoration of burnt sites in Mediterranean forests. Dead woody structures have an advantage over nurse plants because they facilitate regeneration without providing underground competition for resources. Accordingly, these authors declare that this is a promising area of research globally. The studies reported in Chapter 5 and 6 showed that the technique of placing logs on plots after burning or cultivation did not significantly improve or reduce eucalypt germination and seedling survival, although the amount of observed germination of sown seed was correlated with the overall amount of coarse woody debris at a site. Using this element of the seedling recruitment niche as a restoration technique needs further development. It is suggested that coarse woody debris is burnt so that some partially burnt sections remain and are left for some time to build up underlying moisture before sowing seed around them. This technique should more closely resemble the natural regeneration process described in Chapter 3.

Other issues that have arisen from the restoration trials that need further clarification include:

- whether coarse woody debris provides protection from browsing and eliminates the need for fencing (de Chantal and Granstrom 2007);
- how regeneration success is influenced by temporal effects such as time of planting/sowing within and between years (Battaglia 1996; Hobbs and Norton 1996; Vaughn and Young 2010); and

- spatial effects such as treatment patch placement within a remnant (Tongway and Ludwig 1994; Lawrence *et al.* 1998; Hobbs and Cramer 2003).

7.4 Implications of soil water repellency

Unique insights developed in this thesis include the recognition of the importance attached to the spatial and temporal variability of soil water repellency in the eucalypt regeneration niche including its potential to be a barrier to recruitment and its implications for the success of restoration treatments in dry eucalypt woodlands.

A significant finding of this thesis, not previously reported, is that suitable eucalypt seedling microsites are characterised by less water repellent soil in comparison with the surrounding forest floor (Chapter 3). This is consistent with the seedlings being in ashbeds in which heavy fuels were burnt and soils were heated to temperatures that break down surface soil repellency. It is also consistent with seedlings being located in or near preferential pathways or macropores in the soil through which water moves. Further fine scale studies are needed to refine the understanding of these processes.

The fact that soil surrounding lignotuberous sprouts had similar levels of water repellency as the forest floor (with moderate to extreme repellency) suggests that the effects of fire on surface soil repellency in seedling microsites reduces over time. This has also not been reported before in this context. There is need for a more comprehensive study of the temporal variability in water repellency in dry forests and woodlands.

The potential of soil water repellency to act as a barrier to recruitment and as an environmental filter during the recruitment process in dry eucalypt woodlands and

forests was highlighted in Chapters 4 and 5. Experiments following the fates of seedlings in soils with and without inherent water repellency through wetting and drying cycles over time would help elucidate these processes further.

In degraded sites, increased soil water repellency may be a part of the degradation process. This is implied by the restoration study as paired healthy and intermediate sites had different median levels of water repellency, particularly at the Evercreech sites. The understorey at the severely hydrophobic Evercreech plantation site was predominantly pasture grasses and severe water repellency has also been shown to occur in grasslands and pastures overseas (Ritsema and Dekker 1996; Dekker and Ritsema 2000; Doerr *et al.* 2000). The nearby Evercreech coupe site had a healthy and diverse native understorey and predominantly wettable soils. The effects of degrading processes such as grazing and understorey transformation on soil water repellency would be an interesting future line of enquiry that has not previously been considered in the restoration literature.

Soil water repellency was shown to influence the success and growth of planted seedlings in the restoration trial reported in Chapter 6. The negative effect of soil water repellency on the survival of planted seedlings over time was a unique finding. Deaths of seedlings soon after planting and/or during summer were likely to be affected by soil water repellency. This is a major concern for restoration efforts in remnants with inherently water repellent soils. Even if soils do not appear to be hydrophobic at planting they may become so as soils dry out, heightening drought impacts during summer.

Soils need to be tested for water repellency when they are dry, either in the field during summer or after air drying *ex situ* when field soils are wet, in order to

determine if seasonal hydrophobicity is a problem at a particular restoration site. Trials that include the use of soil wetting agents that break down soil repellency, like those undertaken in other ecosystems (Osborn *et al.* 1967; Madsen 2010; Ruthrof *et al.* 2010), may be a useful way of identifying methods to remove this barrier to restoring regeneration.

7.5 Management of remnants for improved regeneration

The requirements for restoration of regeneration differ between remnants of different states (as defined in Figure 7.1) because forests in lower states have passed through additional thresholds and barriers (biotic and abiotic) that influence the success of restoration treatments (Hobbs and Yates 1997).

Healthy sites with prolific regeneration (S1 in Figure 7-1) would not normally need any management to improve regeneration other than keeping degrading influences such as excessive grazing to a minimum. Similarly, healthy sites with adequate regeneration (S2 in Figure 7-1) should be managed to limit degradation, at least in the short to medium term. However in the longer term fire may be needed to be introduced to free up resources, maintain tree health and revitalise the stand (Close *et al.* 2009a).

Careful application of fire to healthy sites with limited recruitment (S3 in Figure 7-1) may also be all that is needed to restore the recruitment niche and induce regeneration to sufficient levels (i.e. make the transition across B1 from S3 to S1 or S2). The effort needed to make artificial wood piles or to undertake cultivation is likely to be unnecessary. At Tom Gibson Reserve at Epping Forest the unburnt section in which the restoration trials were undertaken had a ratio of regenerating stems to adult trees of < 1:1 (0.23:1) while the area studied in Chapter 3 that had

been burnt 3 years prior to the survey had almost ten times more regeneration with a regenerating stem to adult ratio of 2.5:1. Accordingly, the management issues for these types of sites revolves around the timing and safety of the fire event (Forestry Commission 1993), as the climatic conditions before and after a burn influence fire intensity and regeneration success (Chandler *et al.* 1991; Yates *et al.* 1994b).

Management advice for intermediate sites with adequate regeneration (S4 in Figure 7-1) would be to protect regeneration (McElhinny 2005). Control of degrading processes such as grazing/browsing, nutrient enrichment and encroachment of woody weeds (predominantly Gorse, *Ulex europaeus,* and Broom, *Cytisus scoparius,* in the Midlands, Gilfedder *et al.* 2003) may be needed in this type of site to limit damage to, and competition with, existing regeneration.

Three of the trial sites (Bald Hill, Liliesleaf and Valleyfield) were in intermediate health with limited regeneration (S5 in Figure 7-1) and one intermediate site had no regeneration (Evercreech plantation, S6) prior to restoration treatments. Even though results of restoration treatments differed within this group, basic management suggestions can be derived from the trials. Most importantly some form of soil amelioration and thorough weed control need to be undertaken to facilitate regeneration as untreated control plots showed no signs of eucalypt recruitment over two years. In addition, although not directly tested in these trials, protection from browsing and trampling by livestock for at least the first few years after planting/sowing has been shown in other studies to be essential (Yates and Hobbs 1997a; Semple and Koen 2001; Li *et al.* 2003).

The choice of amelioration technique may be influenced by the amount of effort a land manager is able to, or prepared to make. Spot fire burns took a day or two to

prepare and manage and were labour intensive. However, they provided longer term weed control and therefore required less ongoing management. The cultivation technique itself took less than a day to carry out but required multiple visits to undertake weed control. As seedling germination and planted seedling survival were generally superior in cultivated plots, but weed control and relative growth and health of seedlings were generally superior in burnt plots, a combination of cultivation and spot fires in the same plot may provide better results. This is worth testing.

The outcomes of this research confirm conclusions from other studies (Hobbs and Norton 1996; Yates *et al.* 2000a) that a large investment of time and resources is required to force the transition from limited to adequate regeneration in a degraded remnant. Further development of the most suitable range of techniques to be used is needed to inform improved remnant management.

Although there is still need for further refinement of methodology, this research clearly defines the distinction between the recruitment and persistence niches of regeneration in eucalypt woodland and dry forest remnants of Midlands Tasmania. The research also demonstrates beyond doubt that the recreation and indeed the expansion of these critical regeneration niches, using these or similar techniques, are not only possible but essential in the protection and restoration management of these important vegetation remnants.

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